Appendix D3

WAG ERA Exposure Models and Parameter Input Values

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TABLES

Appendix D3

WAG ERA Exposure Models and Parameter Input Values

D3-1. WAG ERA EXPOSURE EQUATIONS AND PARAMETER DATA BASE

Determining exposure and the effects of that exposure on ecological receptors are important parts of the ecological risk assessment (ERA) process and require the compilation of a large information base. Exposure is defined as the contact of a receptor with a contaminant or physical agent. The exposure concentration is the amount (e.g., mg/kg, pCi/g, μg/L) of that contaminant in a medium (i.e., soil, food, water, and air) that the receptor will likely contact. In the risk assessment process, intake or exposure of ecological receptors to contaminants in the environment is generally calculated using basic foodweb models. These intake values are then compared to toxicity reference values (TRVs) to provide an evaluation of the potential effects to receptors.

The purpose of this appendix is to document the exposure equations, receptors (functional groups), input parameters, and TRVs used to assess receptors at the WAG level. The exposure equations presented in the following sections are those applied for all WAG level ERAs and represent preliminary models that will be applied in the OU 10-04 ERA. If, as a result of data reviews outlined on Table C2-1-1, model input values that better reflect site specific conditions can be developed, models will be refined to incorporate those values. For example, models for some receptors could be made more complex if species diet composition, site specific uptake factors or assimilation rates can be gleaned from reviews of INEEL studies and/or sampling. Refinement of exposure models is a data gap that will be filled once OU 10-04 COPCs and receptors have been finalized and ESRF, LDRD and 1997 sampling data have been reviewed for site-specific application. Screening based on final COPC concentrations for OU 10-04 sites may eliminate the need for more detailed modeling for some groups of contaminants (e.g. radionuclides) and receptors. Refinement will then be concentrated on models for receptors and COPC exposures of interest. The appendix compilations are limited to species and contaminants identified as present at the INEEL and all values were specifically derived based on environmental conditions unique to the INEEL.

D3-2. EXPOSURE ASSESSMENT

The major pathways of contaminant exposure to terrestrial receptors at the INEEL are primarily via direct soil ingestion, food chain biotransfer (i.e., consumption of plant and animal matter containing chemicals derived from soil), and surface water ingestion (i.e., consumption of surface water from impoundments and waste ponds). Basic exposure equations using foodweb principles are used to estimate this exposure. The INEEL contaminant exposure may occur from both nonradionuclide and radionuclide contamination. The different exposure equations used to model either exposure to radionuclides or nonradionuclides are presented in the following sections.

D3-2.1 Nonradionuclide Exposure Equations

The following exposure equations for terrestrial receptors are for general exposure (Equation D3-1), for food and soil ingestion exposure (Equation D3-2), and for water ingestion exposure (Equation D3-3).

Equation D3-1 is the general exposure equation for dose in mg/kg body weight-day from soil and/or food and water ingestion.

$$EE_{total} = EE_{soil/food} + EE_{water} \quad \text{(D3-1)}$$

where

EE_{total} = total estimated intake from ingestion of soil, food, and water (mg/kg body weight-day)

$EE_{soil/food}$ = estimated intake from ingestion of food and soil (mg/kg body weight-day)

EE_{water} = estimated intake from ingestion of water (mg/kg body weight-day).

Equation D3-2 is the equation for exposure in mg/kg body weight-day from food and soil ingestion as adapted from EPA's *Wildlife Exposure Factors Handbook* (EPA, 1993).

$$EE_{soil/food} = \frac{[((PP \times BAF) + (PV \times PUF) + (PS)) \times CS] \times IR \times SUF \times ED}{BW} \quad \text{(D3-2)}$$

where

$EE_{soil/food}$ = estimated intake from ingestion of food and/or soil (mg/kg body weight-day)

PP = fraction of diet represented by prey ingested (kg prey/kg diet)

BAF = prey-specific bioaccumulation factor (mg contaminant of potential concern [COPC]/kg animal tissue/mg COPC/kg soil)

PV = fraction of diet represented by vegetation ingested (kg vegetation/kg diet)

PUF = plant uptake factor (mg COPC/kg plant tissue/mg COPC/kg soil).

PS = fraction of diet represented by soil (kg soil/kg diet)

CS = concentration of COPC in soil (mg/kg soil)

IR = total food ingestion rate (kg dry weight/day)

SUF = site use factor (unitless)

ED = exposure duration, i.e., fraction of year spent in the affected area (unitless)

BW = receptor-specific body weight (kg).

Equation D3-3 is the equation for dose in mg/kg body weight-day from surface water ingestion.

$$EE_{water} = CW * WI \quad \text{(D3-3)}$$

where

EE_{water} = estimated intake from ingestion of surface water (mg/kg body weight-day)

CW = contaminant concentration in water (mg/L)

WI = water ingestion rate (L/kg bodyweight-day).

Where water ingestion rate is calculated using the following allometric equations from EPA's *Wildlife Exposures Factors Handbook* (EPA, 1993):

$WI = 0.059\ BW^{0.67}$ *(for birds)*

$WI = 0.099\ BW^{0.90}$ *(for mammals)*

Due to the complexity of water ingestion by reptiles, no general reptilian water ingestion equation is available. It is assumed here that desert reptiles, such as those found at the INEEL, get their water solely from prey.

It is recognized that burrowing and non-burrowing animals are potentially exposed to different soil concentrations. This was modeled by assuming that non-burrowing animals (both predator and prey) are expected to only ingest surface soils; however, burrowing prey was assumed to be exposed to subsurface conditions.

Combining Equations D3-4 and D3-5 gives the following total dose to nonradiological contaminants in mg/kg body weight-day:

for burrowers,

$$EE_{tot} = [(PP \times BAF + PV \times PUF + PS) \times CS_g \times IR + WI \times CW] \left(\frac{ED \times SUF}{BW} \right) \quad \text{(D3-4)}$$

and for non-burrowers,

where

$$EE_{tot} = \{[(PP \times BAF + PV \times PUF) \times CS_g + CS_s \times PS] \times IR + WI \times CW\} \times \left(\frac{ED \times SUF}{BW} \right) \quad \text{(D3-5)}$$

CS_s = surface soil concentration (mg/kg)

CS_g = the greater of the surface and subsurface soil concentrations (mg/kg).

Note that each WAG site was assessed solely for those contaminants that were identified at that location. Generally, surface water and soil/foodweb exposure was evaluated together only for those sites where both types of media were available.

D3-2.2 Radionuclide Exposure Equations

Equations for radionuclides are necessary to adequately model this exposure. Radionuclides can cause either an internal or external dose exposure that must be assessed. All the nonradionuclide and radionuclide equations commonly used at the INEEL for WAG ERAs are presented in the following sections.

$$EE_{radtotal} = EE_{internal\ soil/food} + EE_{internal\ water} + EE_{external}$$

where

$EE_{radtotal}$ = total estimated intake from ingestion of soil, food, and water (pCi/g bodyweight-day)

$EE_{(internal)soil/food}$ = estimated intake from ingestion of food and soil (pCi/g body weight-day)

$EE_{(internal)water}$ = estimated intake from ingestion of water (pCi/g body weight-day)

$EE_{external}$ = estimated external dose from exposure to soil (pCi/g body weight-day).

D3-2.2.1 Internal Radionuclide Dose Rate Equations

Equation D3-6 is used to calculate the internal radiation dose rate estimates by assuming that the steady-state whole body concentration is equivalent to the steady-state concentration of radionuclides in reproductive organs (IAEA 1992).

$$EE_{internal} = \frac{CS \times CF \times ED \times ADE \times FA \times 3200\ dis/day\ -\ pCi}{6.24 \times 10^{9}\ MeV/g\ -\ Gy} \qquad \text{(D3-6)}$$

where

$EE_{(internal)soil/food}$ = internal radiation dose rate estimate (pCi/g body weight-day)

CS = concentration of contaminant in soil (pCi/g)

CF = concentration factor (unitless).

ED = exposure duration (fraction of year spent in the affected area; unitless)

ADE = average decay energy per disintegration (MeV/dis)

FA = fraction of decay energy absorbed (unitless)

Concentration factors (CFs) for radionuclides are discussed in Subsections D3-3.5 and D3-3.6. Assumptions used in the calculation of the average decay energy (ADE) values were (a) for alpha or beta radiation, the FA was set equal to 1 (i.e., 100%), and (b) for gamma radiation, the FA was set equal to 0.3 (i.e., 30%). Only emissions with an intensity of 1% or greater were considered; auger and conversion electrons were not considered. The ADE values for radionuclides are included in Attachment D3-1. The ADE values were calculated using Equation D3-7 (Kocher 1981).

$$ADE = \sum_{i=1}^{n} Y_i E_i \qquad \text{(D3-7)}$$

where

Y_i = yield or intensity

E_i = energy of radiation, for ß = average energy.

Alpha particles are comparatively heavy and have a double charge; they react strongly with matter, producing large numbers of ions per unit length of their path. As a result they are not very penetrating and are usually not hazardous. When internally deposited in the tissue of an organism, however, alpha particles are often more damaging than most other types of radiation because comparatively large amounts of energy are deposited within a very small volume of tissue (Schultz and Whicker, 1982). Therefore, a quality factor of 20 was multiplied times each internal dose calculation to allow for the greater damage possible from internally deposited alpha contamination.

D3-2.2.2 Internal Radiation Dose Rate from Water Ingestion

Water ingestion of radionuclides may occur and will be assessed by using a simple differential equation shown in Equation D3-8.

$$\frac{dTC}{dt} = Intake - \lambda_1(TC) - \lambda_2(TC) - L \qquad \text{(D3-8)}$$

where

TC = tissue concentration (pCi/g tissue)

$Intake$ = intake [(pCi/L)(L/g tissue-day)]

λ_1 = radiological decay constant (1/day)

λ_2 = biological loss constant (1/day)

L = other loss (e.g., through urination) [(pCi/L)(L/g tissue-day)].

Conservatively assuming L = 0 and solving for TC at equilibrium (i.e., dTC/dt = 0) gives Equation D3-9.

$$TC = \frac{Intake}{\lambda_1 + \lambda_2} \qquad \text{(D3-9)}$$

where

TC = tissue concentration (pCi/g tissue)

$Intake$ = intake [(pCi/L)(L/g tissue-day)]

λ_1 = radiological decay constant (1/day)

λ_2 = biological loss constant (1/day).

The daily intake rate of the radionuclide from water is calculated in Equation D3-8.

$$EE_{(\text{internal})\ water} = \frac{CW \times WI}{BW \times 1{,}000\ g/kg} \quad \text{(D3-10)}$$

where

$EE_{(internal)\ water}$ = dose rate estimate (pCi/g body weight-day)

CW = concentration of the radionuclide in water (pCi/L)

WI = water ingestion rate (L/day)

BW = receptor-specific body weight (kg).

D3-2.2.3 External Radiation Dose Rate

External dose is derived using formulas outlined in Shleien (1992). Dose rate-to-tissue in an infinite medium uniformly contaminated by a gamma emitter is calculated using Equation D3-11.

$$EE_{external} = \frac{2.12 \times \overline{E} \times C}{\rho} \quad \text{(D3-11)}$$

where

$EE_{external}$ = external dose rate to tissue (rad/hr)

$\overline{E}$ = average gamma energy per disintegration (MeV/dis)

C = concentration of contaminant ($\mu Ci/cm^3$)

ρ = density of the medium (g/cm^3).

It is not anticipated that external radiation dose from surface water will be a major contributor to risk to terrestrial ecological receptors at the INEEL and was not considered a significant exposure.

D3-3. PARAMETER INPUT VALUES

WAG ERA receptor exposures were calculated using the models presented in Section D3-2 and species-specific input values (PV, PP, PS, IR, WI, BW, ED, SUF). Exposures for each functional group incorporate best estimates to reflect species-specific life history and feeding habits. Defaults and assumptions for selecting soil/sediment and drinking water exposure model input values are given in Table D3-3-1. Finalized parameter input values used to model contaminant intake through consumption of food or water by functional groups and individual species evaluated as part of the WAG ERAs are

presented in Table D3-3-2. These values have been explicitly developed to reflect INEEL contaminant issues. Individual parameter values and literature sources are discussed in the following sections.

D3-3.1 Diet (PV, PP, PS)

Group and individual species diets are represented in the EBSL equations by the sum of three parameters (percent vegetation [PV], percent prey [PP], and percent soil [PS]), constrained to equal 100%. For herbivores, PV is represented by 1–PS, (where PP = 0). No distinction was made between the types of vegetation consumed. Although some species, primarily herbivorous, may consume a small percent of its diet as insect prey, this was considered in the trophic assignment as part of the functional grouping criteria (VanHorn et al. 1995).

Table D3-3-1. Parameter defaults and assumptions applied in WAG ERA dose calculations.

Parameter	WAG ERA Default or Assumption
PV	Herbivores assumed to be 100 - PS Insectivores assumed to be 0 Carnivores is assumed to be 0 Omnivores percent from literature PV - PS/2 + PP - PS/2 + PS
PP	Herbivores assumed to be 0 Insectivores assumed to 100 - PS Carnivores assumed to be 100 - PS Omnivores percent from literature PV - PS/2.
PS	The highest value (i.e., greatest exposure) was selected from species within functional group. Individual species evaluated using values as presented.
IR	Allometric equations (Nagy 1987). The largest IR/BW ratio was used from the species within in functional group.
WI	Allometric equations (EPA 1993). The largest WI/BW ratio was selected from species within each functional group.
BW	The smallest BW/IR ratio was selected from species within each functional group.
ED	The largest value was selected from species within each functional group.
SUF	Calculated as site area divided by species home range area/home range (HR). The largest SUF value was selected from species within each functional group.

Table D3-3-2. Finalized input values for WAG ERA exposure parameters.

Functional Groups	PP	PV	PS	ED	IR (kg/day)	BW (kg)	HR (Ha)	WI (L/day)
Amphibians (A232)	0.941E+01	0.00E+01	5.90E-02	1.00E-00	6.49E-05	8.00E-03	aquatic	0.00E+00
Avian herbivores (AV122)	0.00E+00	9.07E-01	9.30E-02	1.00E-00	1.46E-03	3.50E-03	5.18E-00	1.33E-03
Avian herbivores (AV143)	0.00E+00	9.81E-01	8.20E-02	6.50E-01	2.92E-02	3.47E-01	aquatic	2.90E-02
Trumpeter swan	0.00E+00	9.81E-01	8.20E-02	2.50E-01	2.75E-01	1.09E+01	aquatic	2.93E-01
Avian insectivores (AV210)	9.80E-01	0.00E+00	2.00E-02	6.50E-01	2.90E-03	1.00E-02	8.38E-00	2.70E-03
Black tern	9.80E-01	0.00E+00	2.00E-02	2.50E-01	9.84E-03	6.53E-02	aquatic	9.48E-03
Avian insectivores (AV210A)	9.70E-01	0.00E+00	3.00E-02	6.50E-01	3.89E-03	1.46E-02	2.39E-00	3.48E-03
Avian insectivores (AV222)	9.70E-01	0.00E+00	9.30E-02	1.00E+00	3.07E-03	1.09E-02	aquatic	2.86E-03
Avian insectivores (AV232)	8.20E-01	0.00E+00	1.80E-01	6.50E-01	1.12E-03	2.32E-02	aquatic	1.01E-03
Avian insectivores (AV233)	8.20E-01	0.00E+00	1.80E-01	2.50E-01	4.78E-03	2.12E-02	aquatic	4.50E-03
White-faced ibis	8.90E-01	0.00E+00	1.10E-01	2.50E-01	4.27E-02	6.22E-01	aquatic	4.29E-02
Avian carnivores (AV310)	9.80E-01	0.00E+00	2.00E-02	1.00E-00	1.61E-02	1.39E-01	2.18E+02	1.57E-02
Northern goshawk	9.80E-01	0.00E+00	2.00E-02	2.50E-01	6.00E-02	1.05E-00	2.13E+02	6.10E-02
Peregrine falcon	9.80E-01	0.00E+00	2.00E-02	2.50E-01	4.96E-02	7.82E-01	3.31E+01	5.00E-02
Avian carnivores (AV322)	9.80E-01	0.00E+00	2.00E-02	1.00E-00	7.44E-03	4.25E-02	9.00E-00	7.11E-03
Bald eagle	9.80E-01	0.00E+00	2.00E-02	2.50E-01	1.60E-01	4.74E-00	4.94E+02	1.67E-01
Ferruginous hawk	9.80E-01	0.00E+00	2.00E-02	6.50E-01	6.19E-02	1.10E-00	5.60E+02	6.29E-02
Loggerhead shrike	9.80E-01	0.00E+00	2.00E-02	6.50E-01	7.44E-03	4.25E-02	4.57E-00	7.11E-03
Avian carnivores (AV322A)	9.70E-01	0.00E+00	3.00E-02	2.50E-01	1.73E-02	1.55E-01	1.00E+01	1.69E-02
Burrowing owl	9.70E-01	0.00E+00	3.00E-02	2.50E-01	1.73E-02	1.55E-01	1.00E+01	1.69E-02
Avian omnivores (AV422)	6.27E-01	2.80E-01	9.30E-02	1.00E-00	1.13E-02	8.02E-02	1.10E+01	1.09E-02
Avian omnivores (AV 442)	6.20E-01	2.70E-01	1.10E-01	1.00E+00	4.41E-02	6.54E-01	aquatic	4.44E-02
Mammalian herbivores (M121)	0.00E+00	9.80E-01	2.00E-02	2.50E-01	3.14E-01	5.80E-00	1.10E+01	4.82E-01
Mammalian herbivores (M122)	0.00E+00	9.37E-01	6.30E-02	1.00E-00	3.30E-03	1.10E-02	2.30E-01	1.71E-03
Mammalian herbivores (M122A)	0.00E+00	9.23E-01	7.70E-02	1.00E-00	4.27E-03	1.57E-02	3.00E-01	2.35E-03
Pygmy rabbit	0.00E+00	9.80E-01	2.00E-02	1.00E-00	4.53E-02	4.04E-01	2.80E-01	4.38E-02
Mammalian insectivores (M210)	9.80E-01	0.00E+00	2.00E-02	5.00E-01	1.43E-03	9.03E-03	2.39E-00	1.43E-03
Mammalian insectivores (M210A)	9.80E-01	0.00E+00	2.00E-02	2.50E-01	1.43E-03	4.65E-03	2.39E-00	7.88E-04
Townsend's western big-eared bat	9.90E-01	0.00E+00	1.00E-02	1.00E-00	2.37E-03	1.10E-02	2.39E-00	1.71E-03
Small-footed myotis	9.90E-01	0.00E+00	1.00E-02	1.00E-00	1.44E-03	4.69E-03	2.39E-00	7.94E-04
Long-eared myotis	9.90E-01	0.00E+00	1.00E-02	1.00E-00	1.77E-03	6.65E-03	2.39E-00	1.09E-03
Mammalian insectivores (M222)	9.76E-01	0.00E+00	2.40E-02	1.00E-00	1.66E-03	6.00E-03	1.24E-01	9.91E-04

Table D3-3-2. (continued).

Functional Groups	PP	PV	PS	ED	IR (kg/day)	BW (kg)	HR (Ha)	WI (L/day)
Merriam's shrew	9.76E-01	0.00E+00	2.40E-02	1.00E-00	1.66E-03	6.00E-03	1.24E-01	9.91E-04
Mammalian carnivores (M322)	9.23E-01	0.00E+00	7.70E-02	1.00E-00	1.66E-02	1.78E-01	1.30E+01	2.09E-02
Mammalian omnivores (M422)	8.06E-01	1.00E-01	9.40E-02	1.00E-00	3.06E-03	1.70E-02	7.20E-01	2.53E-03
Reptilian insectivores (R222)	9.76E-01	0.00E+00	2.40E-02	1.00E-00	5.60E-05	6.61E-03	1.17E-01	0.00E+00
Sagebrush lizard	9.76E-01	0.00E+00	2.40E-02	1.00E-00	5.60E-05	6.61E-03	1.17E-01	0.00E+00
Reptilian carnivores (R322)	9.52E-01	0.00E+00	4.80E-02	1.00E-00	6.80E-03	1.50E-02	3.00E-00	0.00E+00
Plants	0.00E+00	0.00E+00	1.00E-00	1.00E-00	NA	NA	NA	NA

NA—not applicable.

For carnivores, PP is represented by 1 – PS, (where PV = 0). Values for the fraction of overall diet represented by prey were taken from species specific or representative species diets as reported in the literature.

Dietary composition for omnivores is represented by (PV – PS/2) + (PP – PS/2) + PS = 1 unless PP or PV are 10% or less, in which case the entire PS value was subtracted from the greater of the two. Dietary profiles for functional groups were based on diets for representative species developed from studies conducted at the INEEL and other regional locations (noted on Table D3-3-3). Since most dietary studies report only in terms of prey or vegetation material, the dietary fraction comprised of soil was evenly subtracted from prey and vegetation fractions of the diet to account for inclusion of ingested soil without exceeding 1. The number of individual species comprising prey was not considered; however, the contribution of prey items to overall diet was based on relative biomass rather than the most numerous individual components. Dietary composition for functional groups is represented by the species having the largest PS within that group.

The values for PS were taken primarily from soil ingestion data presented by Beyer et al. 1994. Species for which values were presented (Beyer et al. 1994) are limited, so soil ingestion values were assigned using professional judgement to match dietary habits with species most similar to INEEL species.

Finalized dietary values and literature sources for functional groups and individual species model for WAG ERAs are presented on Table D3-3-3. Further refinement in the diet of individual species and functional groups is beyond the scope of both screening and WAG-level ERAs. More detailed dietary models will be implemented in the OU 10-04 ERA.

D3-3.2 Body Weight (BW)

Body weights for mammals, amphibians, and reptiles were extracted from numerous local and regional studies. Body weights for birds were taken primarily from Dunning (1993) unless local or regional values were available. Values were chosen in order of preference for study locale: (1) INEEL, (2) Idaho, (3) Regional (sagebrush steppe in Washington, Oregon, Wyoming, Nevada and northern Utah), and (4) U.S.-wide. Where no distinction in sex was reported, mean adult weights were used. In cases where only separate means for male and female were reported, the average of the two was calculated. In cases where only a range in weights could be found, a median value was used. Functional group weight represents the smallest individual species body weight in the group. Finalized body weights for functional groups and individual EBSL calculations and literature sources are given on Table D3-3-4.

D3-3.3 Food and Water Ingestion Rates (IR, WI)

Food/prey ingestion rates (IR) for most INEEL species were calculated using allometric equations given in Nagy (1987). Food intake rates (grams dry weight per day) for passerine birds, nonpasserine birds, rodents, herbivores, all other mammals, and insectivorous reptiles were estimated using the following allometric equations (Nagy 1987).

$$Food\ intake\ rate = 0.398\ BW^{0.850}\ (passerines) \quad \text{(D3-12)}$$

$$Food\ intake\ rate = 1.110\ BW^{0.445}\ (desert\ bird) \quad \text{(D3-13)}$$

$$Food\ intake\ rate = 0.648\ BW^{0.651}\ (all\ birds) \quad \text{(D3-14)}$$

Table D3-3-3. Summary of exposure model input values and literature sources for dietary parameters (PP, PV, and PS).

Functional Groups	PP	PV	PS	PS Model Species[a]
Amphibians (A232)	9.41E-01	0.00E+01	5.90E-02	Eastern painted turtle
Avian herbivores (AV122)	0.00E+01	9.07E-01	9.30E-02	Wild turkey
Avian herbivores (AV143)	0.00E+01	9.18E-01	8.20E-02	Canada goose
Trumpeter swan	0.00E+01	9.18E-01	8.20E-02	Canada goose
Avian insectivores (AV210)	9.80E-01	0.00E+01	2.00E-02	Estimated
Black tern	7.50E-01	0.00E+01	2.00E-02	Estimated
Avian insectivores (AV210A)	9.70E-01	0.00E+01	3.00E-02	Burrowing owl
Avian insectivores (AV222)	9.07E-01	0.00E+01	9.30E-02	Wild turkey
Avian insectivores (AV232)	8.20E-01	0.00E+01	1.80E-01	Western sandpiper
Avian insectivores (AV233)	8.20E-01	0.00E+01	1.80E-01	Western sandpiper
White-faced ibis	8.90E-01	0.00E+01	1.10E-01	Western sandpiper
Avian carnivores (AV310)	9.80E-01	0.00E+01	2.00E-02	Wood duck
Northern goshawk	9.80E-01	0.00E+01	2.00E-02	Estimated
Peregrine falcon	9.80E-01	0.00E+01	2.00E-02	Estimated
Avian carnivores (AV322)	9.80E-01	0.00E+01	2.00E-02	Estimated
Bald eagle	9.80E-01	0.00E+01	2.00E-02	Estimated
Ferruginous hawk	9.80E-01	0.00E+01	2.00E-02	Estimated
Loggerhead shrike	9.80E-01	0.00E+01	2.00E-02	Estimated
Avian carnivores (AV322A)	9.70E-01	0.00E+01	3.00E-02	Burrowing owl
Burrowing owl	9.70E-01	0.00E+01	3.00E-02	Burrowing owl
Avian omnivores[b] (AV422)	6.27E-01	2.80E-01	9.30E-02	Wild turkey
Avian omnivores[b] (AV 442)	6.20E-01	2.70E-01	1.10E-01	Wood duck
Mammalian herbivores (M121)	0.00E+01	9.80E-01	2.00E-02	Mule deer
Mammalian herbivores (M122)	0.00E+01	9.37E-01	6.30E-02	Black-tailed jackrabbit[c]
Mammalian herbivores (M122A)	0.00E+01	9.23E-01	7.70E-02	Black-tailed prairie dog
Pygmy rabbit	0.00E+01	9.80E-01	2.00E-02	Black-tailed prairie dog
Mammalian insectivores[d] (M210)	9.80E-01	0.00E+01	2.00E-02	Beetle specialist
Mammalian insectivores[d] M210A)	9.80E-01	0.00E+01	2.00E-02	Beetle specialist
Townsend's western big-eared bat	9.90E-01	0.00E+01	1.00E-02	Moth specialist
Small-footed myotis	9.90E-01	0.00E+01	1.00E-02	Moth specialist
Long-eared myotis	9.90E-01	0.00E+01	1.00E-02	Beetle specialist
Mammalian insectivores (M222)	9.76E-01	0.00E+01	2.40E-02	Meadow vole

Table D3-3-3. (continued)

Functional Groups	PP	PV	PS	PS Model Species[a]
Merriam's shrew	9.76E-01	0.00E+01	2.40E-02	Meadow vole
Mammalian carnivores (M322)	9.23E-01	0.00E+01	7.70E-02	Black-tailed prairie dog
Mammalian omnivores[e] (M422)	8.06E-01	1.00E-01	9.40E-02	Raccoon
Reptilian insectivores (R222)	9.76E-01	0.00E+01	2.40E-02	Meadow vole
Sagebrush lizard	9.76E-01	0.00E+01	2.40E-02	Meadow vole
Reptilian carnivores (R322)	9.52E-01	0.00E+01	4.80E-02	Fox plus 2%

a. From Beyer et al., 1994, unless otherwise noted.

b. Dietary composition, percent prey and percent vegetation based on avian models from EPA 1993.

c. From Arthur and Gates 1988.

d. Soil ingestion rates for bats were estimated based on primary prey life histories – Beetle strategists = 2% and moth strategists =1%.

e. Dietary composition 90% prey and 10% vegetation based on INEEL data for the coyote (Johnson and Hansen 1979).

Table D3-3-4. Summary of body weight input values for WAG ERA exposure modeling.

Functional Groups	BW (kg)	Representative Species	Reference
Amphibians (A232)	8.00E-03	Boreal chorus frog	Steenhof 1983 (calculated from SVL[a] for spadefoot toads – 0.6 SVL)
Avian herbivores (AV122)	3.50E-03	Rufous hummingbird	Dunning 1993 (mean adult)
Avian herbivores (AV143)	3.47E-01	Cinnamon teal	Steenhof 1983 (mean adult)
Trumpeter swan	1.09E+01	Trumpeter swan	Dunning 1993 (mean adult)
Avian insectivores (AV210)	1.00E-02	Western flycatcher	Dunning 1993 (mean adult)
Black tern	6.53E-02	Black tern	Dunning 1993 (mean adult)
Avian insectivores (AV210A)	1.46E-02	Bank swallow	Dunning 1993 (mean adult)
Avian insectivores (AV222)	1.09E-02	House wren	Dunning 1993 (mean adult)
Avian insectivores (AV232)	2.32E-02	Least sandpiper	Dunning 1993 (mean adult)
Avian insectivores (AV233)	2.15E-02	Willet	Dunning 1993 (mean adult)
White-faced ibis	6.22E-01	White-faced ibis	Dunning 1993 (mean adult)
Avian carnivores (AV310)	1.39E-01	Sharp-shinned hawk	Dunning 1993 (mean adult)
Northern goshawk	1.05E-00	Northern goshawk	Dunning 1993 (mean adult)
Peregrine falcon	7.82E-01	Peregrine falcon	Dunning 1993 (mean adult)
Avian carnivores (AV322)	4.25E-02	Loggerhead shrike	Fraser and Luukkonen 1986 (mean adult)
Bald eagle	4.74E-00	Bald eagle	Dunning 1993 (mean adult)
Ferruginous hawk	1.10E-00	Ferruginous hawk	Steenhof 1993 (mean adult)
Loggerhead shrike	4.25E-02	Loggerhead shrike	Fraser and Luukkonen 1986 (mean adult)
Avian carnivores (AV322A)	1.55E-01	Burrowing owl	Dunning 1993 (mean adult)
Burrowing owl	1.55E-01	Burrowing owl	Dunning 1993 (mean adult)
Avian omnivores (AV422)	8.02E-02	Scrub jay	Dunning 1993 (mean adult)
Avian omnivores (AV442)	6.54E-01	American coot	Steenhof 1983 (mean adult
Mammalian herbivores (M121)	5.80E-00	American porcupine	Steenhof 1983 (mean adult)
Mammalian herbivores (M122)	1.10E-02	Western harvest mouse	Steenhof 1983 (mean adult)
Mammalian herbivores (M122A)	1.57E-02	Sagebrush vole	Mullican 1985 (median adult)
Pygmy rabbit	4.04E-01	Pygmy rabbit	Arthur and Markham 1978 (mean adult)
Mammalian insectivores (M210)	9.03E-03	Silver-haired bat	Barclay et al. 1988 (mean adult)
Mammalian insectivores (M210A)	4.65E-03	California myotis	Black 1974 (mean adult)
Townsend's western big-eared bat	1.10E-02	Townsend's western big-eared bat	Burt and Grossenheider 1964 (median adult)

Table D3-3-4. (continued)

Functional Groups	BW (kg)	Representative Species	Reference
Small-footed myotis	4.69E-03	Small-footed myotis	Barclay et al. 1988 (mean adult)
Long-eared myotis	6.65E-03	Long-eared myotis	Barclay et al. 1988 (mean adult)
Mammalian insectivores (M222)	6.00E-03	Merriam's shrew	Steenhof 1983 (mean adult)
Merriam's shrew	6.00E-03	Merriam's shrew	Steenhof 1983 (mean adult)
Mammalian carnivores (M322)	1.78E-01	Long-tailed weasel	Steenhof 1983 (mean adult)
Mammalian omnivores (M422)	1.70E-02	House mouse	Burt and Grossenheider 1964 (median adult)
Reptilian insectivores (R222)	6.61E-03	Sagebrush lizard	Burkholder 1978 (mean adult)
Sagebrush lizard	6.61E-03	Sagebrush lizard	Burkholder 1978 (mean adult)
Reptilian carnivores (R322)	1.50E-02	Night snake	Steenhof 1983 (mean adult)

a. SVL = snout-to-vent length

$$\textit{Food intake rate} = 0.583\ BW^{0.585}\ \textit{(rodents)} \quad \text{(D3-15)}$$

$$\textit{Food intake rate} = 0.577\ BW^{0.727}\ \textit{(mammalian herbivores)} \quad \text{(D3-16)}$$

$$\textit{Food intake rate} = 0.235\ BW^{0.822}\ \textit{(all other mammals)} \quad \text{(D3-17)}$$

$$\textit{Food intake rate} = 0.15\ BW^{0.874}\ \textit{(desert mammals)} \quad \text{(D3-18)}$$

$$\textit{Food intake rate} = 0.013\ BW^{0.773}\ \textit{(reptile insectivores)} \quad \text{(D3-19)}$$

where BW = body weight in grams.

The original equation for rodents (D3-13) has been modified slightly (Nagy 1987), based on reporting errors discovered in that article. An equation for ingestion rates for carnivorous reptiles was constructed using data reported by Diller and Johnson 1988.

$$\textit{Food intake rate} = 0.01\ BW^{1.6}\ \textit{(reptile carnivores)} \quad \text{(D3-20)}$$

where BW = body weight in kilograms.

These equations were applied to estimate the ingestion rate (g dry weight/day) as a function of body weight. The application of individual equations for species and groups varies according to taxonomic Class and/or Order and in some cases, on habitat (e.g. aquatic species). In cases where more than one of Nagy's (1987) equations could be applied to a functional group, such as all mammals or desert rodents, the larger of the two rates was applied. For functional groups in which mixed species occur, intake rates were calculated using the most representative or generic equation returning the largest IR. Finalized ingestion rates for functional groups and individual species are presented in Table D3-3-5.

Table D3-3-5. Summary of exposure model input values and equations for calculation of food and water (IR, WI) ingestion for groups and individuals.

Functional Groups	IR (kg/day)	Nagy Equation	WI[b] (L/day)
Amphibians (A232)	6.49E-05	reptile insectivores	0.00E+00
Avian herbivores (AV122)	1.46E-03	all birds	1.33E-03
Avian herbivores (AV143)	2.92E-02	all birds	2.90E-02
Trumpeter swan	2.75E-01	all birds	2.93E-01
Avian insectivores (AV210)	2.90E-03	all birds	2.70E-03
Black tern	9.84E-03	all birds	9.48E-03
Avian insectivores (AV210A)	3.89E-03	Passerines	3.48E-03
Avian insectivores (AV222)	3.07E-03	all birds	2.86E-03
Avian insectivores (AV232)	1.12E-03	all birds	1.01E-03
Avian insectivores (AV233)	4.78E-03	all birds	4.50E-03
White-faced ibis	4.27E-02	all birds	4.29E-02
Avian carnivores (AV310)	1.61E-02	all birds	1.57E-02
Northern goshawk	6.00E-02	all birds	6.10E-02
Peregrine falcon	4.96E-02	all birds	5.00E-02
Avian carnivores (AV322)	7.44E-03	all birds	7.11E-03
Bald eagle	1.60E-01	all birds	1.67E-01
Ferruginous hawk	6.19E-02	all birds	6.29E-02
Loggerhead shrike	7.44E-03	all birds	7.11E-03
Avian carnivores (AV322A)	1.73E-02	all birds	1.69E-02
Burrowing owl	1.73E-02	all birds	1.69E-02
Avian omnivores (AV422)	1.13E-02	all birds	1.09E-02
Avian omnivores (AV442)	4.41E-02	all birds	4.44E-02
Mammalian herbivores (M121)	3.14E-01	mammal herbivore	4.82E-01
Mammalian herbivores (M122)	3.30E-03	mammal herbivore	1.71E-03
Mammalian herbivores (M122A)	4.27E-03	mammal herbivore	2.35E-03
Pygmy rabbit	4.53E-02	mammal herbivore	4.38E-02
Mammalian insectivores (M210)	1.43E-03	Rodents	1.43E-03
Mammalian insectivores (M210A)	1.43E-03	Rodents	7.88E-04
Townsend's western big-eared bat	2.37E-03	Rodents	1.71E-03
Small-footed myotis	1.44E-03	Rodents	7.94E-04
Long-eared myotis	1.77E-03	Rodents	1.09E-03
Mammalian insectivores (M222)	1.66E-03	Rodents	9.91E-04
Merriam's shrew		Rodents	9.91E-04
Mammalian carnivores (M322)	1.66E-02	all mammals	2.09E-02
Mammalian omnivores (M422)	3.06E-03	Rodents	2.53E-03
Reptilian insectivores (R222)	5.60E-05	reptile insectivores	0.00E+00
Sagebrush lizard	5.60E-05	reptile insectivores	0.00E+00
Reptilian carnivores (R322)	6.80E-03	literature value[a]	0.00E+00

a. Diller and Johnson 1988.

b. Calculated using EPA 1993 allometric equations.

A cursory comparison of food ingestion values generated using Nagy's equations to a few experimental values from the literature indicate that the equations may substantially underestimate ingestion rates for some species.

Water ingestion rates were calculated for functional groups and individual species using dry diet allometric equations for birds and mammals (EPA 1993). Reptiles and amphibians were assumed to attain water through absorption and metabolic processes. Although other species (some birds and small mammals) meet water needs through metabolic and dietary means, these species were assumed to ingest water for drinking based on the equations. Allometric equations used in calculating water ingestion rates for individual species and functional groups are presented below.

Water ingestion for individual species was found from the following equations (EPA 1993):

$$WI = 0.059\ BW^{0.67}\ \textit{(for all birds)} \qquad \text{(D3-21)}$$

$$WI = 0.099\ BW^{0.90}\ \textit{(for all mammals)} \qquad \text{(D3-22)}$$

where body weight is in units of kg.

Finalized water ingestion rates for functional groups and individual species evaluated for the WAG ERAs are presented on Table D3-3-5.

D3-3.4 Exposure Duration (ED)

Exposure duration (ED) represents the fraction of year an animal spends in the affected area. For WAG level exposure calculations, ED was modified to reflect species migratory patterns. Previously identified status and abundance data compiled for INEEL species (Arthur et al. 1984) were used to assign individual species to one or more of five status/abundance categories including: resident, breeding, summer visitor, migratory, and winter visitor. An estimated minimum and maximum percentage of the year individuals may spend on-Site was assigned for each category, where resident species = 0.05 to 1.00 (birds, migratory and transient mammals), species breeding on site = 0.05 to 0.65, and migratory, summer and winter visitors—0.05 to 0.25. (in terms of days). For example, ED is assumed to be 1 for year-round residents (i.e., receptors spend 100% of their time in the assessment area). For migratory receptors spending only one season (e.g., summer or winter) onsite, ED is assumed to be 0.50. Maximum range values were used in all cases, and exposure durations for functional groups are represented by the largest ED value across species within each functional group. Finalized ED values for functional groups and individual species analyzed as part of the WAG ERA assessments are listed on Table D3-3-6.

D3-3.5 Site Use Factor (SUF) and Home Range (HR)

The site use factor (SUF) represents the proportion of a species home range that overlaps the area of contamination. An SUF of 1 indicates that the home range is less than or equal to the area of contaminant exposure. Exposure estimates were corrected for site size in the WAG ERA models. The SUF was calculated for groups or individual species by dividing the assessment or site area by the species (or functional group) home range. Aquatic receptors assessed for the WAG ERAs are assumed to be limited to individual sites having surface water (e.g. sewage disposal ponds), so were assigned a SUF of 1. Home range values were taken primarily from the literature and were compared to calculated home

Table D3-3-6. Summary of exposure duration input values for WAG ERA exposure modeling.

Functional Groups	ED
Amphibians (A232)	1.00E-00
Avian herbivores (AV122)	1.00E-00
Avian herbivores (AV143)	6.50E-01
Trumpeter swan	2.50E-01
Avian insectivores (AV210)	6.50E-01
Black tern	2.50E-01
Avian insectivores (AV210A)	6.50E-01
Avian insectivores (AV222)	1.00E+00
Avian insectivores (AV232)	6.50E-01
Avian insectivores (AV233)	2.50E-01
White-faced ibis	2.50E-01
Avian carnivores (AV310)	1.00E-00
Northern goshawk	2.50E-01
Peregrine falcon	2.50E-01
Avian carnivores (AV322)	1.00E-00
Bald eagle	2.50E-01
Ferruginous hawk	6.50E-01
Loggerhead shrike	6.50E-01
Avian carnivores (AV322A)	2.50E-01
Burrowing owl	2.50E-01
Avian omnivores (AV422)	1.00E-00
Avian omnivores (AV442)	1.00E-00
Mammalian herbivores (M121)	2.50E-01
Mammalian herbivores (M122)	1.00E-00
Mammalian herbivores (M122A)	1.00E-00
Pygmy rabbit	1.00E-00
Mammalian insectivores (M210)	5.00E-01
Mammalian insectivores (M210A)	2.50E-01
Townsend's western big-eared bat	1.00E-00
Small-footed myotis	1.00E-00
Long-eared myotis	1.00E-00
Mammalian insectivores (M222)	1.00E-00
Merriam's shrew	1.00E-00
Mammalian carnivores (M322)	1.00E-00
Mammalian omnivores (M422)	1.00E-00
Reptilian insectivores (R222)	1.00E-00
Sagebrush lizard	1.00E-00
Reptilian carnivores (R322)	1.00E-00

ranges generated using allometric equations (McNab 1963; Harestad and Bunnell 1979; Turner et al. 1969; Linstedt et al. 1986). Where calculated values were substantially smaller than literature values, the calculated values were used. For groups and species for which no literature values could be found, allometric equations or extrapolations from other species were used. Home range values compiled for functional groups and individual species, along with associated references, are shown on Table D3-3-7.

D3-3.6 Transfer Factors (BAF, PUF)

The uptake of contaminants in the terrestrial food chain is important for realistically calculating exposure to contamination. These contaminant-specific factors are referred to in the literature as uptake factors or plant uptake factors (PUFs) for plants and food-chain transfer coefficients or factors for wildlife. The PUF is the plant tissue concentration of the contaminant divided by the soil or sediment concentration. The food-chain transfer factor is the animal tissue concentration of a contaminant divided by the concentration in its food. These factors will be developed first from site-specific data or from the general literature, if possible. Data on chemical concentrations in wild animals, as opposed to domestic or laboratory animals is limited in the literature. Hanford has produced the *Ecotoxicity Literature Review of Selected Hanford Site Contaminants* (Driver 1994). This report states that food chain transport information is generally lacking for desert or sagebrush-steppe organisms. It has shown that these values tend to be highly site-specific, due to the effects that biological and physiochemical factors may have on contaminant bioavailability and toxicity. It is important to note that use of literature values adds considerable uncertainty to the assessment.

To estimate the tissue levels of contaminants in prey items of wildlife, the PUF was multiplied by the transfer factors to derive a "bioaccumulation factor" (BAF), which is the concentration of a contaminant in the tissues of an animal divided by the soil or sediment concentration. The BAF accounts for all ingestion exposure routes. For example, the BAF for a herbivorous small mammal is the PUF times the plant-to-herbivore transfer coefficient. Multiplying the small mammal BAF times the concentration of a contaminant in soil provides an estimate of the tissue levels of the contaminant in small mammals. This tissue level may then be used to estimate exposure for the carnivore/omnivore functional groups that are predators of small mammals. However, it is noted that BAFs appropriate for the INEEL should be calculated and used to support focused exposure and risk assessment (*e.g.*, Clifford et al. 1995) at the second and third phase of the ERA process.

In the exposure analysis, equation D3-22 will yield the concentration of contaminant in the prey.

$$CP = CS \times BAF \qquad \text{(D3-22)}$$

where

CP = concentration in prey item ingested (mg/kg)

CS = concentration of contaminant in soil (mg/kg)

BAF = contaminant-specific bioaccumulation factor (unitless).

The concentration of contaminant in vegetation (CV) can be estimated using equation D3-23:

Table D3-3-7. Finalized home range values for WAG ERA exposure parameters.

Functional Groups	HR (Ha)	Representative Species	Reference
Amphibians (A232)	area of site	Aquatic	N/A
Avian herbivores (AV122)	5.18E-00		
Avian herbivores (AV143)	area of site	Aquatic	N/A
Trumpeter swan	area of site	Aquatic	N/A
Avian insectivores (AV210)	8.38E-00	Eastern kingbird	Mace and Harvey 1983
Black tern	8.38E-00	Eastern kingbird	Mace and Harvey 1983
Avian insectivores (AV210A)	2.39E-00	Horned lark	Mace and Harvey 1983; Schoener 1968
Avian insectivores (AV222)	area of site	Aquatic	N/A
Avian insectivores (AV232)	area of site	Aquatic	N/A
Avian insectivores (AV233)	area of site	Aquatic	N/A
White-faced ibis	area of site	Aquatic	N/A
Avian carnivores (AV310)	2.18E+02	Calculated	Harestad and Bunnell 1979; Mace and Harvey 1983
Northern goshawk	2.13E+02	Northern goshawk	Mace and Harvey 1983
Peregrine falcon	3.31E+01	Calculated	Harestad and Bunnell 1979
Avian carnivores (AV322)	9.00E-00	~2 times Loggerhead shrike	Mace and Harvey 1983; Harestad and Bunnell 1979
Bald eagle	4.94E+02	Snowy owl	Schoener 1968
Ferruginous hawk	5.60E+02	Ferruginous hawk	Gerhardt 1989
Loggerhead shrike	4.57E-00	Loggerhead shrike	Mace and Harvey 1983
Avian carnivores (AV322A)	1.00E+01	0.5 of Burrowing owl calculated value	Harestad and Bunnell 1979
Burrowing owl	1.00E+01	0.5 of calculated	Harestad and Bunnell 1979
Avian omnivores (AV422)	1.10E+01	Scrub jay – calculated	Harestad and Bunnell 1979
Avian omnivores (AV442)	area of site	aquatic	N/A
Mammalian herbivores (M121)	1.10E+01	Porcupine	Harestad and Bunnell 1979
Mammalian herbivores (M122)	2.30E-01	Western harvest mouse	McNab 1963
Mammalian herbivores (M122A)	3.00E-01	Pygmy rabbit - calculated	Green and Flinders 1980

Table D3-3-7. (continued).

Functional Groups	HR (Ha)	Representative Species	Reference
		calculated	
Mammalian insectivores (M210)	2.39E-00	Chipping sparrow	Mace and Harvey 1983
Mammalian insectivores (M210A)	2.39E-00	Chipping sparrow	Mace and Harvey 1983
Townsend's western big-eared bat	2.39E-00	Chipping sparrow	Mace and Harvey 1983
Small-footed myotis	2.39E-00	Chipping sparrow	Mace and Harvey 1983
Long-eared myotis	2.39E-00	Chipping sparrow	Mace and Harvey 1983
Mammalian insectivores (M222)	1.24E-01	Merriam's shrew-calculated	McNab 1963 – cropper
Merriam's shrew	1.24E-01	Merriam's shrew-calculated	McNab 1963 – cropper
Mammalian carnivores (M322)	1.30E+01	Long-tailed weasel - calculated	Harestad and Bunnell 1979
Mammalian omnivores (M422)	7.20E-01	Deer mouse	Koehler 1988
Reptilian insectivores (R222)	1.17E-01	Sagebrush lizard	Guyer 1978
Sagebrush lizard	1.17E-01	Sagebrush lizard	Guyer 1978
Reptilian carnivores (R322)	3.00E-00	Gopher snake – calculated	Turner et al. 1969

$$CV = CS \times PUF \qquad \text{(D3-23)}$$

where

CV = concentration in vegetation (mg/kg)

CS = concentration of contaminant in soil (mg/kg)

PUF = contaminant-specific plant uptake factor (unitless).

For discussion of food-chain transfer factors for radionuclides, it is more appropriate to refer to a concentration factor (CF). However, the same values are used since biouptake is not related to radioactivity. CFs are referred to as appropriate.

Lacking actual site-specific data for organisms at the INEEL, an effort has been made to select BAFs that would be protective in any environmental situation. The basic approach was to examine the available literature for those COPCs that have been shown to bioaccumulate. In view of the many gaps in our knowledge of chemical biotransfer in the terrestrial environment, non-site-specific BAFs are necessarily highly uncertain. This method for developing BAFs should be used with care and for situations where there are established steady-state conditions. Attachment D3-2 to this appendix contains a discussion of the BAFs developed for the INEEL and used in the ERAs. The BAFs for organics are estimated using the Travis and Arms (1988) equation (log BAF = -7.735 + 1.033 log K_{ow}). Log

partitioning coefficients (K_{ow}s) were taken from Montgomery and Welkom (1990). BAFs are shown in Table D3-3-8.

Site-specific PUFs should also be used whenever possible. However, these were not available for the WAG ERAs, PUFs for all metals are taken from Baes et al. (1984) as discussed in Attachment D3-3. Commonly, the element specific PUFs available from Baes et al. (1984) are used in the risk assessment. Baes et al. (1984) give preference to studies that reported the steady-state concentration of metals in plants at edible maturity, various soil properties are not considered, and data for numerous plant species (both animal feeds and those consumed by humans) are combined. Since root uptake is a complex process that depends on various soil properties (e.g., pH, cation exchange capacity, and organic matter content), as well as the element and type of plant involved. The PUF for organics contaminants was estimated using the geometric mean regression equation developed by Travis and Arms (1988), where the log of PUF = 1.588 - 0.578 log K_{ow}. Log partitioning coefficients (K_{ow}s) were selected from Montgomery and Welkom (1990). Tables D3-3-9 and D3-3-10 present the PUFs for inorganics and organics respectively.

There is a great deal of uncertainty associated with the BAFs used to calculate dose. Very few BAFs are available in the scientific literature, since they must be both contaminant- and receptor-specific. In the absence of specific BAFs, a value of 1 was assumed. This assumption could over- or underestimate the true dose from the contaminant, and the magnitude of error cannot be quantified. Travis and Arms (1988) and Baes et al. (1984) report BAFs for contaminants to beef and milk; all of these are less than 1 for the contaminants at the INEEL. If the terrestrial receptors of concern accumulate metals and PCBs in a similar way and to a comparable degree as beef and dairy cattle, the use of a BAF of 1 for all contaminants and receptors would overestimate the dose. On the other hand, if the terrestrial receptors of concern at the INEEL accumulate metals and PCBs to a much larger degree than beef and dairy cattle, the assumption of BAFs equal to 1 could underestimate the true dose from the COPCs.

D3-4. TOXICITY REFERENCE VALUE (TRV) DEVELOPMENT

Toxicity assessment consists of hazard evaluation, and dose-response assessment. The hazard evaluation involves a comprehensive review of toxicity data for COPCs to identify the nature and severity of toxic properties, especially with respect to key receptors or similar species. Dose-response assessment would allow the prediction of the amount of chemical exposure that might result in adverse ecological effects. No dose-based toxicological criteria were available for ecological receptors at the INEEL. Therefore, it was necessary to develop appropriate toxicity reference values (TRVs) for the COPCs and receptors at the INEEL. A TRV is defined as a dose for a receptor taxon (including sensitive subgroups such as taxa under regulatory protection) that is likely to be without appreciable risk of deleterious effects from chronic exposure.

The development of TRVs for the INEEL is discussed in detail in Appendix D4. Table D3-4-1 and D3-4-2 present the TRVs used at the INEEL for ERA purposes.

Table D3-3-8. Concentration factors BAFs (or CFs) for INEEL contaminants (unitless).

Contaminants	BAF[a] for Insectivores[b]	BAF for Carnivores[c]	BAF for Omnivores[d]
Inorganics[e]			
Antimony	9.0E-01	6.0E-03	9.0E-01
Arsenic	1.0E+00	4.0E-02	1.0E+00
Barium	1.0E+00	1.5E-01	1.0E+00
Cadmium	1.1E+00	1.9E+00	1.9E+00
Chromium III	6.0E-02	2.0E-01	2.0E-01
Chromium VI	6.0E-02	2.0E-01	2.0E-01
Copper	1.0E+00	2.0E-01	1.0E+00
Fluoride	1.0E+00	1.0E+00	1.0E+00
Lead	3.0E-01	6.0E-01	6.0E-01
Mercury	4.0E-01	7.0E-01	7.0E-01
Selenium	1.0E+00	3.2E-01	1.0E+00
Silver	1.0E+00	4.0E-01	1.0E+00
Strontium[f]	1.5E+00	1.5E+00	1.5E+00
Thallium	1.0E+00	4.0E-02	1.0E+00
Tin	1.0E+00	8.0E-02	1.0E+00
Zinc	1.0E+00	7.0E-01	1.0E+00
Organics[g]			
1,1-Dichloroethylene	1.6E-06	1.6E-06	1.6E-06
1,1,1 Trichloroethane	7.9E-06	7.9E-06	7.9E-06
1,1,2-Trichloro-1,2,2-Trifluoroethane	2.5E-06	2.5E-06	2.5E-06
1,2,4-Trichlorobenzene	5.0E-04	5.0E-04	5.0E-04
1,1,2,2-Tetrachloroethane	6.2E-06	6.2E-06	6.2E-06
1,4-Dichlorobenzene	1.0E-04	1.0E-04	1.0E-04
2-Butanone	4.6E-08	4.6E-08	4.6E-08
2-Chlorotoluene	4.8E-05	4.8E-05	4.8E-05
2-Hexanone	1.0E+00	1.0E+00	1.0E+00
2-Methylnaphthalene	3.0E-04	3.0E-04	3.0E-04
2-Nitrophenol	1.4E-06	1.4E-06	1.4E-06
2-Propanol	1.0E+00	1.0E+00	1.0E+00
2,3,7,8,-Tetrachloro dibenzodioxin	1.0E+00	1.0E+00	1.0E+00
2,4- Dimethylphenol	1.0E+00	1.0E+00	1.0E+00
2,4-Dichlorophenoxyacetic acid	1.6E-05	1.6E-05	1.6E-05
2,4-Dinitrotoluene	2.5E-06	2.5E-06	2.5E-06
4-Chloroaniline	1.7E-06	1.7E-06	1.7E-06
4-Methylphenol	2.3E-06	2.3E-06	2.3E-06
4-Chloro-3-methylphenol (CMP)	1.0E+00	1.0E+00	1.0E+00
Acenaphthene	2.5E-04	2.5E-04	2.5E-04

Table D3-3-8. (continued).

Contaminants	BAF[a] for Insectivores[b]	BAF for Carnivores[c]	BAF for Omnivores[d]
Acetone	1.4E-08	1.4E-08	1.4E-08
Acetonitrile	1.0E+00	1.0E+00	1.0E+00
Acrylonitrile	1.0E+00	1.0E+00	1.0E+00
Anthracene	7.1E-04	7.1E-04	7.1E-04
Benzene	3.3E-06	3.3E-06	3.3E-06
Benzine	1.0E+00	1.0E+00	1.0E+00
Benzo(a)anthracene	1.0E-02	1.0E-02	1.0E-02
Benzo(a)pyrene	2.9E-02	2.9E-02	2.9E-02
Benzo(b)fluoranthene (BbF)	2.9E-02	2.9E-02	2.9E-02
Benzo(k)fluoranthene	2.9E-02	2.9E-02	2.9E-02
Benzo(g,h,i)perylene	8.1E-02	8.1E-02	8.1E-02
Butyl alcohol	1.0E+00	1.0E+00	1.0E+00
Butylbenzylphthalate (BBP)	1.6E-03	1.6E-03	1.6E-03
Carbon disulfide	2.5E-06	2.5E-06	2.5E-06
Carbon tetrachloride	1.7E-05	1.7E-05	1.7E-05
Chloroform	2.3E-06	2.3E-06	2.3E-06
Chloromethane (Methyl chloride)	2.4E-08	2.4E-08	2.4E-08
Chrysene	1.0E-02	1.0E-02	1.0E-02
Cyanide	1.0E+00	1.0E+00	1.0E+00
Decanal	1.0E+00	1.0E+00	1.0E+00
Dibenzofuran	1.0E+00	1.0E+00	1.0E+00
Dichlorodifluoromethane	3.6E-06	3.6E-06	3.6E-06
Di-2-ethylhexyl-phthalate (DEHP)	2.4E-04	2.4E-04	2.4E-04
Diethyl phthalate	7.9E-06	7.9E-06	7.9E-06
Dimethyl phthalate	3.3E-06	3.3E-06	3.3E-06
Di-n-butylphthalate	1.0E-02	1.0E-02	1.0E-02
Di-n-octylphthalate	4.0E+01	4.0E+01	4.0E+01
Ethanol (Ethyl alcohol)	1.2E-08	1.2E-08	1.2E-08
Ethylbenzene	3.5E-05	3.5E-05	3.5E-05
Fluoranthene	2.0E-03	2.0E-03	2.0E-03
Fluorene	4.0E-04	4.0E-04	4.0E-04
Formaldehyde	2.5E-08	2.5E-08	2.51E-08
Hydrazine	2.1E-11	2.1E-11	2.1E-11
Isophorone	1.3E-06	1.3E-06	1.3E-06
Indeno(1,2,3)pyrene	7.9E-02	7.9E-02	7.9E-02
Mercury (Organic)	1.0E+00	1.0E+00	1.0E+00
Methanol (Methyl alcohol)	1.0E+00	1.0E+00	1.0E+00

Table D3-3-8. (continued).

Contaminants	BAF[a] for Insectivores[b]	BAF for Carnivores[c]	BAF for Omnivores[d]
Methly isobutyl ketone	1.0E+00	1.0E+00	1.0E+00
Methylene chloride	5.0E-07	5.0E-07	5.0E-07
n-Propylbenzene	1.0E+00	1.0E+00	1.0E+00
Naphthalene	6.9E-05	6.9E-05	6.9E-05
Orthophosphate	1.0E+00	1.0E+00	1.0E+00
PCBs - Aroclor 1254	2.7E-02	2.7E-02	2.7E-02
PCBs - Aroclor 1260	3.5E-01	3.5E-01	3.5E-01
PCBs	2.8E-02	2.8E-02	2.8E-02
Phenanthrene	7.2E-04	7.2E-04	7.2E-04
Phenol	7.2E-07	7.2E-07	7.2E-07
Propionitrile	1.0E+00	1.0E+00	1.0E+00
Pyrene	1.9E-03	1.9E-03	1.9E-03
Sodium cyanide	1.0E+00	1.0E+00	1.0E+00
Sulfide	1.0E+00	1.0E+00	1.0E+00
Sulfuric acid	1.0E+00	1.0E+00	1.0E+00
Terphenyl	1.0E+00	1.0E+00	1.0E+00
Tetrachloroethylene	1.0E-05	1.0E-05	1.0E-05
Tetrahydrofuran	1.0E+00	1.0E+00	1.0E+00
Toluene	1.4E-05	1.4E-05	1.4E-05
Total Petroleum Hydrocarbon	1.0E+00	1.0E+00	1.0E+00
Tributyl phosphate	1.0E+00	1.0E+00	1.0E+00
Trichloroethylene (Trichloroethene)	6.0E-06	6.0E-06	6.0E-06
Trimethylopropane-triester	1.0E+00	1.0E+00	1.0E+00
Vinyl acetate	1.0E+00	1.0E+00	1.0E+00
Xylene (mixed)	4.6E-05	4.6E-05	4.6E-05

a. Bioconcentration factor.
b. BAFs or CFs for insectivores, appropriate for AV200 and M200 Functional Groups.
c. BAFs or CFs for carnivorous, appropriate for AV300 and M300 Functional Groups.
d. BAFs or CFs for omnivores, appropriate for AV400 and M400 Functional Groups.
e. Values and/or literature (Attachment D3-2) for inorganics are from Baes et al., (1984).
f. Site-specific data (VanHorn et al., 1995).
g. Values for organics are from allometric equations presented in Travis and Arms (1988).

Table D3-3-9. Plant uptake factors suggested for native INEEL plants for inorganics.

Contaminant of Concern	Suggested for Native INEEL Plants	Contaminant of Concern	Suggested for Native INEEL Plants	Contaminant of Concern	Suggested for Native INEEL Plants
Aluminum (Al)	4.0E-03	Hydrogen (H)	0.0	Rubidium (Rb)	9.0E-01
Americium (Am)	1.7E-02	Iodine (I)	3.4E-03	Ruthenium (Ru)	2.0E-01
Antimony (Sb)	2.0E-01	Iron (Fe)	4.0E-03	Selenium (Se)	2.5E-02
Arsenic (As)	4.0E-02	Lanthanum (La)	1.0E-02	Silver (Ag)	4.0E-01
Barium (Ba)	1.5E-01	Lead (Pb)	2.0E-02	Sodium (Na)	3.0E-01
Beryllium (Be)	1.0E-02	Manganese (Mn)	9.8E+00	Strontium (Sr)	1.9E+01
Cadmium (Cd)	5.5E-01	Mercury (Hg)	9.0E-01	Technicium (Tc)	9.5E+00
Calcium (Ca)	3.5E+00	Molybdenum (Mo)	8.0E-01	Tellunium (Te)	7.0E+00
Carbon (C)	1.0E+00	Neodymium (Nd)	2.0E-02	Tin (Sn)	3.0E-02
Cerium (Ce)	3.0E-02	Niobium (Nb)	5.0E-02	Thallium (Tl)	4.0E-03
Cesium (Cs)	5.3E-01	Nickel (Ni)	6.0E-02	Thorium (Th)	3.9E-02
Chloride (Cl)	7.0E+01	Neptunium (Np)	1.0E-01	Tungsten (W)	4.5E-02
Chromium (Cr)	1.9E-01	Plutonium (Pu)	4.4E-03	Uranium (U)	1.4E-02
Cobalt (Co)	1.1E+01	Polonium (Po)	7.0E-02	Vanadium (V)	5.5E-03
Copper (Cu)	8.0E-01	Praseodymium (Pr)	2.0E-02	Yttrium (Y)	1.0E-02
Curium (Cm)	1.3E-03	Radium (Ra)	2.6E-02	Zinc (An)	3.5E+01
Europium (Eu)	1.0E-02	Rhodium (Rh)	9.0E-01	Zirconium (Sr)	1.0E-03

Table D3-3-10. Plant uptake factors calculated for organic COPC at the INEEL.

	PUF		PUF
1,1-Dichloroethane	3.57E+00	Carbon tetrachloride	1.15E+00
1,1-Dichloroethylene	3.35E+00	Cerium chloride	1.00E+00
1,1,1-trichloroethane	1.39E+00	Chloroform	2.81E+00
1,1,2,2-Tetrachloroethane	1.61E+00	Cyanide	1.00E+00
1,2 Dichloroethane	5.40E+00	Diethyl phthalate	1.39E+00
1,2,4-Trichlorobenzene	1.26E-01	Di-2-ethylhexylphthalate	1.94E-01
1,3 Dinitrobenzene	4.48E+00	Di-n-butylphthalate	2.25E-02
1,4 Dioxane	3.83E+01	Di-n-octylphthalate	1.87E-04
2-Butanone	2.74E+01	Ethanol	5.93E+01
2-Chlorotoluene	4.11E-01	Ethylbenzene	5.86E-01
2-Propanol	1.00E+00	Fluoranthene	5.70E-02
2,3,7,8-Tetrachlorodibenzodioxin	5.06E-03	Fluorene	1.45E-01
2,4-Dichlorophenoxyacetic acid	9.20E-01	Formaldehyde	3.87E+01
2,4-Dimethylphenol	1.55E+00	Hydrazine	2.33E+03
2,4-Dinitrotoluene	2.70E+00	Mercury(Organic)	9.00E-01
2,6-Dinitrotoluene	2.70E+00	Methanol	1.00E+00
4-Chloroaniline	3.39E+00	Methyl isobutyl ketone	1.00E+00
4-Methylphenol	2.97E+00	Methylene chloride	6.85E+00
4-Chloro-3-methylphenol	7.23E-01	Naphthalene	3.97E-01
Acenaphthene	1.89E-01	Nitrobenzene	3.30E+00
Acetone	5.33E+01	Pentachloronitrobenzene	2.74E-02
Acetonitrile	6.09E+01	Pentachlorophenol	1.00E+00
Acrylonitrile	2.77E+01	Phenanthrene	1.02E-01
Anthracene	1.04E-01	Phenol	5.55E+00
Aroclor 1254	1.27E-02	Pyrene	5.85E-02
Aroclor 1260	2.93E-03	Sulfuric acid	1.00E+00
Benzene	2.30E+00	Terphenyl	1.00E+00
Benzine	1.00E+00	Tetrachloroethylene	1.22E+00
Benzo(a)anthracene	2.25E-02	Toluene	1.02E+00
Benzo(a)pyrene	1.22E-02	Tributyl phosphate	1.00E+00
Butyl Alcohol	1.00E+00	Trichloroethylene	1.63E+00
Butylbenzylphthalate	6.51E-02	Trimethylolpropane-triester	1.00E+00
Carbon disulfide	2.70E+00	Xylene	5.04E-01

Table D3-4-1. Summary of selected toxicity reference values (TRVs in mg/kg-day) for mammalian functional group.

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
1,1-Dichloroethylene (Dog-NOAEL)	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	3.1	3.1
1,1,1 Trichloroethane (Mouse - NOAEL)	333	333	333	333	333	333	333	333	500	500
1,1,2,2-Tetrachloroethane (Mouse - FEL)	7.89	7.89	7.89	7.89	7.89	7.89	7.89	7.89	11.8	11.8
1,2-Dichloroethane (Mouse - NOAEL)	17	17	17	17	17	17	17	17	25	25
1,2,4-Trichlorobenzene (Rat - NOAEL)	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.83	0.83
1,3-Dinitrobenzene (Rat - NOAEL)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.2	0.2
1,4-Dioxane (Rat - NOAEL)	0.17	0.17	0.17	0.17	0.17	0.17	0.17	0.17	0.25	0.25
2-Butanone (Rat - NOAEL)	295.2	295.2	295.2	295.2	295.2	295.2	295.2	295.2	442.8	442.8
2-Chlorotoluene (Rat - NOAEL)	3.3	3.3	3.3	3.3	3.3	3.3	3.3	3.3	5.0	5.0
2-Methylnaphthalene [a] (use benzo(a)pyrene values)	a	a	a	a	a	a	a	a	a	a
2-Propanol (Mouse and rat - NOAEL)	217	217	217	217	217	217	217	217	325	325
2,3,7,8-Tetrachlorodibenzodioxin (Rat - NOAEL)	3E-7	3E-7	3E-7	3E-7	3E-7	3E-7	3E-7	3E-7	5E-7	5E-7
2,4-Dichlorophenoxyacetic acid (Rat and Mouse - NOAEL)	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.5	0.5
2,4- Dimethylphenol (Mouse - NOAEL)	17	17	17	17	17	17	17	17	25	25
2,4-Dinitrotoluene (Mouse - LOAEL)	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.8	1.8
2,4,6 - Trinitrotoluene (Mouse - NOAEL)	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	2.5	2.5
2,6 - Dinitrotoluene (Beagle Dog - LOAEL)	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	2.5	5.0

Table D3-4-1. (continued).

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
4-Chloroaniline (Rat and Mouse - LOAEL)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.78	0.78
4-Methylphenol (p-Cresol) (Rat - NOAEL)	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	6.3	6.3
4-Chloro-3-methylphenol (CMP) (Rat - NOAEL)	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	8.3	8.3
Acenaphthene (Mouse - NOAEL)	14.6	14.6	14.6	14.6	14.6	14.6	14.6	14.6	21.9	21.9
Acetone (Mouse and Rat - NOAEL)	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	13	13
Acetonitrile (Rat - NOAEL)	5.3	5.3	5.3	5.3	5.3	5.3	5.3	5.3	7.9	7.9
Acrylonitrile (Rat - NOAEL)	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.14	0.14
Aluminum (Bovine - NOAEL)	2.55	2.55	2.55	2.55	1.70	1.70	1.70	1.70	1.70	1.70
Aluminum chloride (Bovine - NOAEL)	4.82	4.82	4.82	4.82	3.21	3.21	3.21	3.21	3.21	3.21
Aluminum hydroxide (Rat - AEL)	12.3	12.3	12.3	12.3	12.3	12.3	12.3	12.3	18.5	18.5
Aluminum nitrate (Rat - LOAEL)	15	15	15	15	15	15	15	15	23	23
Ammonia (Rat - LD_{50})	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	2.2	2.2
Anthracene (Mouse - NOAEL)	41.7	41.7	41.7	41.7	41.7	41.7	41.7	41.7	62.5	62.5
Antimony (Mouse - LOAEL)	0.417	0.417	0.417	0.417	0.417	0.417	0.417	0.417	0.625	0.625
Arsenic (Rat - NOAEL)	0.26	0.26	0.26	0.26	0.26	0.26	0.26	0.26	0.39	0.39
Asbestos (Mouse - NOAEL)	67	67	67	67	67	67	67	67	100	100
Barium (Rat - NOAEL)	3.4	3.4	3.4	3.4	3.4	3.4	3.4	3.4	5.1	5.1
Barium chloride (Rat - NOAEL)	5.81	5.81	5.81	5.81	5.81	5.81	5.81	5.81	8.71	8.71
Benzene (Mouse - LOAEL)	3.66	3.66	3.66	3.66	3.66	3.66	3.66	3.66	5.49	5.49
Benzine[b] (Mouse - LD_{50})	b	b	b	b	b	b	b	b	b	b
Benzo(a)anthracene (Mouse - FEL)	9.3	9.3	9.3	9.3	9.3	9.3	9.3	9.3	14	14
Benzo(a)pyrene (Mouse-LOAEL)	0.83	0.83	0.83	0.83	0.83	0.83	0.83	0.83	1.3	1.3

Table D3-4-1. (continued).

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
Benzo(b)fluoranthene (BbF)[a] (use benzo(a)pyrene values)	a	a	a	a	a	a	a	a	a	a
Beryllium (Rat - NOAEL)	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.33	0.33
Bis(tri-n-butyltin)oxide (Mouse – NOAEL)	7.8	7.8	7.8	7.8	7.8	7.8	7.8	7.8	11.7	11.7
Boron (Rat - NOAEL)	2.92	2.92	2.92	2.92	2.92	2.92	2.92	2.92	4.38	4.38
Butyl alcohol (n-Butanol) (Rat - NOAEL)	41.7	41.7	41.7	41.7	41.7	41.7	41.7	41.7	62.5	62.5
Butylbenzylphthalate (BBP) (Rat - NOAEL)	4.42	4.42	4.42	4.42	4.42	4.42	4.42	4.42	6.63	6.63
Cadmium (Rat - LOAEL)	8E-4	8E-4	8E-4	8E-4	8E-4	8E-4	8E-4	8E-4	1E-3	1E-3
Carbon disulfide[c] (Rabbit and Rat - NOAEL)	c	c	c	c	c	c	c	c	c	c
Carbon tetrachloride (Rat - NOAEL)	3.3	3.3	3.3	3.3	3.3	3.3	3.3	3.3	5.0	5.0
Cerium chloride (Rat - FEL)	8.68	8.68	8.68	8.68	8.68	8.68	8.68	8.68	13.03	13.03
Chloroform (Rat - NOAEL)	12.5	12.5	12.5	12.5	12.5	12.5	12.5	12.5	18.8	18.8
Chromium (III) (Rat – NOAEL)	250	250	250	250	250	250	250	250	375	375
Chromium (VI) (Dog – NOAEL)	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.08	0.15
Chrysene[d] (use benzo(a)pyrene values)	d	d	d	d	d	d	d	d	d	d
Cobalt (Dog - NOAEL)	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	2.1	4.2
Copper (Mink - NOAEL)	0.65	0.65	0.65	0.65	0.65	0.65	0.65	1.95	0.65	0.65
Cyanide (Rat - NOAEL)	1.80	1.80	1.80	1.80	1.80	1.80	1.80	1.80	2.70	2.70
Diethylphthalate (Rat – NOAEL)	62.5	62.5	62.5	62.5	62.5	62.5	62.5	62.5	93.8	93.8
Di-2-ethylhexyl-phthalate (DEHP) (Guinea Pig – LOAEL)	0.79	0.79	0.79	0.79	0.79	0.79	0.79	0.79	1.2	1.2
Di-n-butylphthalate (Rat - NOAEL)	4.63	4.63	4.63	4.63	4.63	4.63	4.63	4.63	6.94	6.94
Di-n-octylphthalate (Rat - LOAEL)	14.5	14.5	14.5	14.5	14.5	14.5	14.5	14.5	21.8	21.8

Table D3-4-1. (continued).

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
Ethanol (Rat - LOAEL)	2.66	2.66	2.66	2.66	2.66	2.66	2.66	2.66	3.99	3.99
Ethylbenzene (Rat - LOAEL)	17.0	17.0	17.0	17.0	17.0	17.0	17.0	17.0	25.5	25.5
Fluoranthene (Mouse - NOAEL)	10.4	10.4	10.4	10.4	10.4	10.4	10.4	10.4	15.6	15.6
Fluorene (Mouse – LOAEL)	10.4	10.4	10.4	10.4	10.4	10.4	10.4	10.4	15.6	15.6
Fluoride (Mink – NOAEL)	10.5	10.5	10.5	10.5	10.5	10.5	10.5	31.4	10.5	10.5
Formaldehyde (Rat - NOAEL)	5.0	5.0	5.0	5.0	5.0	5.0	5.0	5.0	7.5	7.5
Hexachlorobenzene (Rat - NOAEL)	6.7	6.7	6.7	6.7	6.7	6.7	6.7	6.7	10	10
HMX (Rat - NOAEL)	17	17	17	17	17	17	17	17	25	25
Hydrazine (Mouse - NOAEL)	0.93	0.93	0.93	0.93	0.93	0.93	0.93	0.93	1.4	1.4
Hydrofluoric acid[c] (Mouse - FEL)	c	c	c	c	c	c	c	c	c	c
Lead (Rat - NOAEL)	2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7	4.0	4.0
Manganese (Rat - NOAEL)	29	29	29	29	29	29	29	29	44	44
Mercury (Inorganic) (Mouse - NOAEL)	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.17	0.17
Mercury (Organic) (Mouse - NOAEL)	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.04	0.04
Methanol (Rat - NOAEL)	170	170	170	170	170	170	170	170	250	250
Methly isobutyl ketone (Rat - NOAEL)	21	21	21	21	21	21	21	21	31	31
Methylene chloride (Rat - NOAEL)	1.95	1.95	1.95	1.95	1.95	1.95	1.95	1.95	2.93	2.93
Molybdenum (Guinea pig - LOAEL)	3.3	3.3	3.3	3.3	3.3	3.3	3.3	3.3	5.0	5.0
n-Propylbenzene[e] (use benzene values)	e	e	e	e	e	e	e	e	e	e
Naphthalene (Mouse - NOAEL)	0.44	0.44	0.44	0.44	0.44	0.44	0.44	0.44	0.66	0.66
Nickel (Dog - NOAEL)	19.0	19.0	19.0	19.0	19.0	19.0	19.0	19.0	28.5	57.0

Table D3-4-1. (continued).

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
Nitrate (Guinea pig - NOAEL)	169	169	169	169	169	169	169	169	254	254
Nitric acid[c] (Rat - NOAEL)	c	c	c	c	c	c	c	c	c	c
Nitrobenzene (Rat - FEL)	1.85	1.85	1.85	1.85	1.85	1.85	1.85	1.85	2.78	2.78
PCBs - Aroclor 1254 (Rat - NOAEL)	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.16	0.16
PCBs - Aroclor 1260 (Rat - NOAEL)	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	3.7	3.7
Pentachlorophenol (Rat - NOAEL)	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.06	0.06
Phenol (Rat - NOAEL)	13	13	13	13	13	13	13	13	20	20
Potassium chloride (Mouse - LD_{50})	6.2	6.2	6.2	6.2	6.2	6.2	6.2	6.2	9.3	9.3
Potassium hydroxide (Rat - LD_{50})	0.51	0.51	0.51	0.51	0.51	0.51	0.51	0.51	0.76	0.76
Potassium nitrate (Guinea pig - NOAEL)	169	169	169	169	169	169	169	169	254	254
Potassium phosphate (Rat - LD_{50})	5.8	5.8	5.8	5.8	5.8	5.8	5.8	5.8	8.6	8.6
Potassium sulfate (Rat - LD_{90})	10.0	10.0	10.0	10.0	10.0	10.0	10.0	10.0	15.0	15.0
Pyrene (Mouse - NOAEL)	13	13	13	13	13	13	13	13	19	19
RDX (Rat - NOAEL)	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.15	0.15
Selenium (Rat - NOAEL)	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.20	0.20
Silver (Swine - NOAEL)	11.3	11.3	11.3	11.3	11.3	11.3	11.3	11.3	17.0	17.0
Sodium chloride (Rat - FEL)	2.88	2.88	2.88	2.88	2.88	2.88	2.88	2.88	4.33	4.33
Sodium hydroxide (Rabbit - LD_{50})	3.1	3.1	3.1	3.1	2.1	2.1	2.1	2.1	2.1	2.1
Sodium phosphate (Mouse - LD_{50})	16.1	16.1	16.1	16.1	16.1	16.1	16.1	16.1	24.2	24.2
Strontium (Rat - NOAEL)	32	32	32	32	32	32	32	32	48	48
Sulfate (Rat - LD_{90})	5.32	5.32	5.32	5.32	5.32	5.32	5.32	5.32	7.98	7.98

Table D3-4-1. (continued).

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
Sulfuric acid[c] (Guinea pig - LOAEL)	c	c	c	c	c	c	c	c	c	c
Terphenyl (Rat - LOAEL)	2.3	2.3	2.3	2.3	2.3	2.3	2.3	2.3	3.5	3.5
Tetrachloroethylene (Mouse - NOAEL)	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.8	1.8
Tetrahydrofuran[f] (use 2,3,7,8-Tetrachlorodibenzodioxin values)	f	f	f	f	f	f	f	f	f	f
Tetryl (Rabbit - LOAEL)	16.3	16.3	16.3	16.3	10.8	10.8	10.8	10.8	10.8	10.8
Thallium (Rat - LOAEL)	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.06	0.06
Tin (Rat - NOAEL)	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.72	1.72
Toluene (Rat - NOAEL)	18.6	18.6	18.6	18.6	18.6	18.6	18.6	18.6	27.9	27.9
Total Petroleum Hydrocarbon[e] (use benzene values)	e	e	e	e	e	e	e	e	e	e
Tributyl phosphate (Rat - LD_{50})	12.3	12.3	12.3	12.3	12.3	12.3	12.3	12.3	18.5	18.5
Trichloroethylene (Rat - NOAEL)	8.3	8.3	8.3	8.3	8.3	8.3	8.3	8.3	13	13
Trimethylopropane-triester (Rat - LD_{50})	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.06	0.06
Uranium (Mouse - NOAEL)	0.512	0.512	0.512	0.512	0.512	0.512	0.512	0.512	0.768	0.768
Vanadium (Mouse - NOAEL)	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.68	0.68
Xylene (Mouse - NOAEL)	0.086	0.086	0.086	0.086	0.086	0.086	0.086	0.086	0.129	0.129
Zinc (Rat - NOAEL)	14	14	14	14	14	14	14	14	21	21
Zirconium (Mouse - NOAEL)	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.87	0.87

a. Use values for benzo(a)pyrene.

b. Exposure route: intravenous injection, not appropriate for ingestion pathway. See Appendix D4 Attachment 2A.

c. Exposure route: inhalation. See Appendix D4 Attachment 2A.

d. Only study available evaluates dermal exposure. See Appendix D4 Attachment 2A.

e. . Use benzene values.

f. Values for 2,3,7,8 –Tetrachlorodibenzodioxin were used.

Table D3-4-1. (continued).

Chemical	TRV for M121	TRV for M122	TRV for M122A	TRV for M123	TRV for M210	TRV for M210A	TRV for M222	TRV for M322	TRV for M422	TRV for M422A
No data were located for:										
	trans-1,3-dichloropropene									
4-nitrophenol	benzo(g,h,i)perylene									
benzoic acid	pentachlorophenol									
phenanthrene	sulfide									

Table D3-4-2. Summary of selected toxicity reference values (TRVs in mg/kg-day) for avian functional groups.

Chemical	TRV for AV121	TRV for AV122	TRV for AV132	TRV for AV142	TRV for AV143	TRV for AV210	TRV for AV210A	TRV for AV221	TRV for AV222	TRV for AV222A	TRV for AV232
1,2-Dichloroethane (Chicken -NOAEL)	2.9	2.9	2.9	2.9	2.9	2.9	2.9	2.9	2.9	2.9	2.9
2,3,7,8,- Tetrachloro dibenzodioxin[a] (Pheasant - NOAEL)	a	a	a	a	a	a	a	a	a	a	a
Aluminum (Chicken -NOAEL)	75	75	75	75	75	75	75	75	75	75	75
Aluminum hydroxide (Chicken - NOAEL)	292	292	292	292	292	292	292	292	292	292	292
Aluminum sulfate (Turkey - NOAEL)	13.5	13.5	13.5	13.5	13.5	13.5	13.5	13.5	13.5	13.5	13.5
Arsenic (Mallard - NOAEL)	0.64	0.64	0.64	1.29	1.29	0.43	0.43	0.43	0.43	0.43	0.43
Bis(tri-n-butyltin)oxide (Japanese Quail – NOAEL)	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1
Boron (Mallard - NOAEL)	14.4	14.4	14.4	28.8	28.8	9.6	9.6	9.6	9.6	9.6	9.6
Cadmium (Black Duck - LOAEL)	0.04	0.04	0.04	0.07	0.07	0.02	0.02	0.02	0.02	0.02	0.02
Chromium-III (Chicken - NOAEL)	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4
Cobalt (Chicken - LOAEL)	0.213	0.213	0.213	0.213	0.213	0.213	0.213	0.213	0.213	0.213	0.213
Copper (Chicken - NOAEL)	4.61	4.61	4.61	4.61	4.61	4.61	4.61	4.61	4.61	4.61	4.61
Cyanide (European Starling - LD_{50})	0.06	0.06	0.06	0.06	0.06	0.08	0.17	0.08	0.08	0.08	0.08
Fluoride (Screech Owl - NOAEL)	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3
Lead (Chicken - NOAEL)	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48
Manganese (Chicken – NOAEL)	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0

Table D3-4-2. (continued).

Chemical	TRV for AV121	TRV for AV122	TRV for AV132	TRV for AV142	TRV for AV143	TRV for AV210	TRV for AV210A	TRV for AV221	TRV for AV222	TRV for AV222A	TRV for AV232
Mercury (Inorganic) (Chicken - NOAEL)	2.02	2.02	2.02	2.02	2.02	2.02	2.02	2.02	2.02	2.02	2.02
Mercury (Organic) (Mallard - LOAEL)	0.004	0.004	0.004	0.008	0.008	0.003	0.003	0.003	0.003	0.003	0.003
Nickel (Mallard - NOAEL)	50	50	50	100	100	33	33	33	33	33	33
Nitrate (Turkey - FEL)	8.9	8.9	8.9	8.9	8.9	8.9	8.9	8.9	8.9	8.9	8.9
PCBs (1254) (Pheasant - LOAEL)	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08
Pentachloronitrobenzene (Chicken - NOAEL)	1.18	1.18	1.18	1.18	1.18	1.18	1.18	1.18	1.18	1.18	1.18
Selenium (Mallard - NOAEL)	0.13	0.13	0.13	0.25	0.25	0.08	0.08	0.08	0.08	0.08	0.08
Silver (Turkey - LOAEL)	14.6	14.6	14.6	14.6	14.6	14.6	14.6	14.6	14.6	14.6	14.6
Sulfate (Turkey - NOAEL)	8.64	8.64	8.64	8.64	8.64	8.64	8.64	8.64	8.64	8.64	8.64
Thallium (Quail - FEL)	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
Uranium (Black Duck - NOAEL)	20	20	20	40	40	13	13	13	13	13	13
Vanadium (Mallard – NOAEL)	5.7	5.7	5.7	11.4	11.4	3.8	3.8	3.8	3.8	3.8	3.8
Zinc (Chicken - LOAEL)	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0

Table D3-4-2. (continued).

Chemical	TRV for AV233	TRV for AV241	TRV for AV242	TRV for AV310	TRV for AV322	TRV for AV333	TRV for AV342	TRV for AV422	TRV for AV432	TRV for AV433	TRV for AV442
1,2-Dichlorobenzene (Chicken - NOAEL)	2.9	2.9	2.9	2.9	2.9	2.9	2.9	4.3	4.3	4.3	4.3
2,3,7,8,-Tetrachloro dibenxodioxin[a] (Pheasant - NOAEL)	a	a	a	a	a	a	a	a	a	a	a
Aluminum (Chicken - NOAEL)	75	75	75	75	75	75	75	112.5	112.5	112.5	112.5
Aluminum hydroxide (Chicken - NOAEL)	292	292	292	292	292	292	292	438	438	438	438
Aluminum sulfate (Turkey - NOAEL)	13.5	13.5	13.5	13.5	13.5	13.5	13.5	20.3	20.3	20.3	20.3
Arsenic (Mallard - NOAEL)	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43
Bis(tri-n-butyltin)oxide (Japanese Quail - NOAEL)	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.7	1.7	1.7	1.7
Boron (Mallard - NOAEL)	9.6	9.6	9.6	9.6	9.6	9.6	9.6	9.6	9.6	9.6	9.6
Cadmium (Black Duck - LOAEL)	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Chromium-III (Chicken - NOAEL)	1.4	1.4	1.4	1.4	1.4	1.4	1.4	2.0	2.0	2.0	2.0
Cobalt (Chicken - LOAEL)	0.213	0.213	0.213	0.213	0.213	0.213	0.213	0.319	0.319	0.319	0.319
Copper (Chicken - NOAEL)	4.61	4.61	4.61	4.61	4.61	4.61	4.61	6.91	6.91	6.91	6.91
Cyanide (European Starling - LD_{50})	0.08	0.08	0.08	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06
Fluoride (Screech Owl - NOAEL)	1.3	1.3	1.3	2.0	2.0	2.0	2.0	1.3	1.3	1.3	1.3
Lead (Chicken - NOAEL)	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.72	0.72	0.72	0.72
Manganese (Chicken - NOAEL)	70.0	70.0	70.0	70.0	70.0	70.0	70.0	105	105	105	105
Mercury (Inorganic) (Chicken - NOAEL)	2.02	2.02	2.02	2.02	2.02	2.02	2.02	3.03	3.03	3.03	3.03
Mercury (Organic) (Mallard - LOAEL)	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003
Nickel (Mallard - NOAEL)	33	33	33	33	33	33	33	33	33	33	33
Nitrate (Turkey - FEL)	8.9	8.9	8.9	8.9	8.9	8.9	8.9	13.4	13.4	13.4	13.4

Table D3-4-2. (continued).

Chemical	TRV for AV233	TRV for AV241	TRV for AV242	TRV for AV310	TRV for AV322	TRV for AV333	TRV for AV342	TRV for AV422	TRV for AV432	TRV for AV433	TRV for AV442
PCBs (1254) (Pheasant - LOAEL)	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.11	0.11	0.11	0.11
Selenium (Mallard - NOAEL)	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08
Pentachloronitrobenzene (Chicken - NOAEL)	1.18	1.18	1.18	1.18	1.18	1.18	1.18	1.77	1.77	1.77	1.77
Silver (Turkey - LOAEL)	14.6	14.6	14.6	14.6	14.6	14.6	14.6	21.8	21.8	21.8	21.8
Sulfate (Turkey - NOAEL)	8.64	8.64	8.64	8.64	8.64	8.64	8.64	12.96	12.96	12.96	12.96
Thallium (Quail - FEL)	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.07	0.07	0.07	0.07
Uranium (Black Duck - NOAEL)	13	13	13	13	13	13	13	13	13	13	13
Vanadium (Mallard - NOAEL)	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8
Zinc (Chicken - LOAEL)	2.0	2.0	2.0	2.0	2.0	2.0	2.0	3.0	3.0	3.0	3.0

a. Exposure route: intraperitoneal injection. See Appendix D4 Attachment 2A.

No data located for:

2-butanone
4-nitrophenol
anthracene
asbestos
barium
benzo(a)anthracence
benzo(a)pyrene
benzo(b)fluoranthene
benzo(g,h,i)perylene
benzo(k)fluoranthene
benzoic acid
butylbenzylphthalate
chloride
chrysene
di-2-ethylhexyl-phthalate
di-n-butylphthalate
di-n-octylphthalate
diethyl phthalate
fluoranthene
methylene chloride
pentachlorophenol
phenanthrene
pyrene

Table D3-4-2. (continued).

Chemical	TRV for AV233	TRV for AV241	TRV for AV242	TRV for AV310	TRV for AV322	TRV for AV333	TRV for AV342	TRV for AV422	TRV for AV432	TRV for AV433	TRV for AV442
2,3,7,8,-Tetrachloro dibenxodioxin[c] (*Pheasant - NOAEL*)	c	c	c	c	c	c	c	c	c	c	c
Aluminum (Chicken - NOAEL)	75	75	75	75	75	75	75	112.5	112.5	112.5	112.5
Aluminum hydroxide (Chicken - NOAEL)	292	292	292	292	292	292	292	438	438	438	438
Aluminum sulfate (Turkey - NOAEL)	13.5	13.5	13.5	13.5	13.5	13.5	13.5	20.3	20.3	20.3	20.3
Arsenic (*Mallard - NOAEL*)	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43
Bis(tri-n-butyltin)oxide (Japanese Quail - NOAEL)	5.0	5.0	5.0	5.0	5.0	5.0	5.0	7.5	7.5	7.5	7.5
Boron (Mallard - NOAEL)	100	100	100	100	100	100	100	100	100	100	100
Cadmium (Black Duck - LOAEL)	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*
Chromium-III (Chicken - NOAEL)	1.4	1.4	1.4	1.4	1.4	1.4	1.4	2.0	2.0	2.0	2.0
Cobalt (Chicken - *LOAEL*)	*0.213*	*0.213*	*0.213*	*0.213*	*0.213*	*0.213*	*0.213*	*0.319*	*0.319*	*0.319*	*0.319*
Copper (Chicken - NOAEL)	4.61	4.61	4.61	4.61	4.61	4.61	4.61	6.91	6.91	6.91	6.91
Cyanide (European *Starling - LD_{50}*)	0.06	0.06	0.06	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Fluoride (Screech Owl - NOAEL)	1.3	1.3	1.3	2.0	2.0	2.0	2.0	1.3	1.3	1.3	1.3
Lead (European *Starling - LOAEL*)	0.04	0.04	0.04	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
Manganese (Chicken - NOAEL)	70.0	70.0	70.0	70.0	70.0	70.0	70.0	105	105	105	105
Mercury (Inorganic) (*Japanese quail - NOAEL*)	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.24	0.24	0.24	0.24
Mercury (Organic) (Mallard - LOAEL)	0.008	0.008	0.008	0.008	0.008	0.008	0.008	0.008	0.008	0.008	0.008
Nickel (Chicken - NOAEL)	2.1	2.1	2.1	2.1	2.1	2.1	2.1	3.1	3.1	3.1	3.1
Nitrate (Turkey - FEL)	8.9	8.9	8.9	8.9	8.9	8.9	8.9	13.4	13.4	13.4	13.4

Table D3-4-2. (continued).

Chemical	TRV for AV233	TRV for AV241	TRV for AV242	TRV for AV310	TRV for AV322	TRV for AV333	TRV for AV342	TRV for AV422	TRV for AV432	TRV for AV433	TRV for AV442
PCBs (1254) (Pheasant - LOAEL)	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.11	0.11	0.11	0.11
Selenium (Mallard - NOAEL)	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08
Sodium nitrate (Juvenile turkeys – LOAEL)	18.3	18.3	18.3	18.3	18.3	18.3	18.3	27.5	27.5	27.5	27.5
Sulfate (Turkey - NOAEL)	8.64	8.64	8.64	8.64	8.64	8.64	8.64	12.96	12.96	12.96	12.96
Thallium (Quail - FEL)	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.07	0.07	0.07	0.07
Uranium (Black Duck - NOAEL)	13	13	13	13	13	13	13	13	13	13	13
Vanadium (Mallard - NOAEL)	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08
Zinc (Chicken - LOAEL)	2.0	2.0	2.0	2.0	2.0	2.0	2.0	3.0	3.0	3.0	3.0

a. Exposure route: inhalation.

b. Exposure route: dermal.

c. Exposure route: intraperitoneal injection.

d. Exposure route: intravenous injection.

No data located for:

2-butanone
4-nitrophenol
anthracene
asbestos
barium
benzo(a)anthracence
benzo(a)pyrene
benzo(b)fluoranthene
benzo(g,h,i)perylene
benzo(k)fluoranthene
benzoic acid
boron
butylbenzylphthalate
chloride
chrysene
di-2-ethylhexyl-phthalate
di-n-butylphthalate
di-n-octylphthalate
diethyl phthalate
fluoranthene
methylene chloride
nitrates
pentachlorophenol
phenanthrene
pyrene

D3-5. REFERENCES

Arthur, W. J. and O. D. Markham, 1978, "Ecology studies at the Idaho National Engineering Laboratory Radioactive Waste Management Complex," In: Markham, O. D. editor, *Ecological Studies on the Idaho National Engineering Laboratory Site 1978 Progress Report,* IDO-12087, U. S. Department of Energy, Idaho Operations Office, Idaho Falls, ID.

Arthur, W. J. and R. J. Gates, 1988, "Trace Element Intake via Soil Ingestion in Pronghorns and in Black-Tailed Jackrabbits," *Journal Range Management* 41:162–166.

Arthur, W. J., J. W. Connelly, D. K. Halford, and T. D. Reynolds, 1984, *Vertebrates of the Idaho National Engineering Laboratory, Idaho Falls, ID,* DOE/ID-12099, U. S. Department of Energy, Idaho Operations Office, pp. 40.

Baes, C. F., III, R. D. Sharp, A. Sjoreen, and R. Shor, 1984, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*, ORNL-5786, U.S. Department of Energy, Oak Ridge National Laboratory, Oak Ridge, TN.

Barclay, R. M. R., P. A. Faure, and D. R. Farr, 1988, "Roosting Behavior and Roost Selection by Migrating Silver-haired Bats," *Journal of Mammal*, 69(4):821–825.

Beyer, W. N., E. E. Connor and D. Gerould, 1994, "Estimates of Soil Ingestion by Wildlife," *Journal of Wildlife Management*, 58:375–382.

Black, H. L., 1974, "A North Temperate Bat Community; Structure and Prey Populations," *Journal of Mammals*, 55(1):138–157.

Burkholder, G. L., 1973, *Life History and Ecology of the Great Basin Sagebrush Swift, Sceloporous graciosus* Baird and Girard, 1852, PhD. Dissertation: Brigham Young University, Salt Lake City, UT.

Burt, W. H. and R. P. Grossenheider, 1980, *A Field Guide to the Mammals.*

Clifford, 1995,

Diller, L.V. and D. R. Johnson, 1988, "Food Habits, Consumption Rates and Predation Rates of Western Rattlesnakes and Gopher Snakes in Southwestern Idaho," *Herpetologia,* 44(2):228–233.

Driver, C. J., 1994, *Ecotoxicity Literature Review of Selected Hanford Site Contaminants.*

Dunning, J. P., 1993, *CRC Handbook of Avian Body Masses*, CRC Press, Boca Raton, FL.

EPA, 1993, *Wildlife Exposure Factors Handbook,* Volumes I and II, EPA/600/R-93/187a and EPA/600/R-93/187b, Office of Research and Development, Washington, D.C.

Fraser, J. D. and D. R. Luukkonen, 1986, "The Loggerhead Shrike," *Audubon Wildlife Report,* p. 933-941.

Gerhardt, 1989

Green, J. S. and J. T. Flinders, 1980, *Brachylagus idahoensis,* Mammalian Species 125:1–4.

Guyer, C., 1978, "Home Range and Homing Ability in *Sceloporus graciosus* and *Phrynosoma douglasii,"* M. S. Thesis: Idaho State Univ., Pocatello, ID, p. 130.

Harestad, A. S. and F. L. Bunnell, 1979, Home Range and Body Weight-a Reevaluation, *Ecology* 60(2):389–402.

IAEA, (International Atomic Energy Agency), 1992, *Effects of Ionizing Radiation on Plants and Animals at Levels Implied by current Radiation Standards*, Technical Report Series No. 332.

Johnson, M. K. and R. M. Hansen, 1979, "Coyote Food Habits on the Idaho National Engineering Laboratory," *Journal Wildlife Management*, 43:951-956.

Kocher, 1981, *Radioactive Decay Data Tables*, NTIs DOE-TIC-11026.

Koehler, D. K., 1988, *Small Mammal Movement Patterns Around a Radioactive Waste Disposal Area in Southeastern Idaho*, M.S. Thesis: University of Wyoming, Laramie, WY, p. 95.

Linstedt, S. L., B. J. Miller, and S. W. Buskirk, 1986, Home Range, Time, and Body Size in Mammals, Ecology 67(2):413–418.

Mace, G. M. and P. H. Harvey, 1983, "Energetic Constraints on Home-Range Size," *Am. Nat.*, 121(1):120-131.

McNab, B. K, 1963, "Bioenergetics and the Determination of Home Range Size," *Am. Nat.*, 97:133–140.

Montgomery and Welkom, 1990, *Groundwater Chemicals Desk Reference*, Lewis Publishers, Chelsea, MI.

Mullican, T. R., 1985, *Ecology of the Sagebrush Vole (Lemmiscus curtatus) in Southeastern Idaho*, M.S. Thesis: Idaho State University, Pocatello, ID.

Nagy, K. A., 1987, "Field Metabolic Rate and food requirement Scaling in mammals and Birds," *Ecological Mono,* 57:111–128.

Schoener, T. W., 1968, "Sizes of Feeding Territories Among Birds," *Ecology,* 49(1):123–141.

Schultz V. and F. W. Whicker, 1982, *Radioecology: Nuclear Energy and the Environment*, CRC Press, Inc., Boca Raton, FL.

Shleien, B., 1992, *The Health Physics and Radiological Health Handbook*, Scinta Publishers, Silver Springs, MD.

Steenhof, K., 1983, "Prey Weights for Computing Percent Biomass in Raptor Diets," *Raptor Research,* 17(1):17–27.

Travis, C. C., and A. D. Arms, 1988, "Bioconcentration of Organics in Beef, Milk, and Vegetation," *Environmental Science and Technology*, 22:271–274.

Turner, F. B., R. I. Jennrich and J. D. Weintraub,1969, "Home Ranges and Body Size of: Lizards," *Ecology*, 50(6):1076–1081.

VanHorn, R. L., N. L. Hampton, and R. C. Morris, 1995, *Draft Guidance Manual for Conducting Screening Level Ecological Risk Assessment at the INEL*, Idaho National Engineering Laboratory, EG&G Idaho, Inc., Idaho Falls, ID, INEL-95/0190, April 1995.

Appendix D3
Attachment 1

Average Decay Energy Table

CONTENTS

TABLE

Table D3-1-1. ADEs for radionuclides at the INEEL.[a]

Radionuclide	ADE (MeV/dis)		
	Alpha	Beta	Gamma
Ac-225	5.75E+00	0.00E+00	1.01E-02
Ac-227	0.00E+00	9.50E-03	1.23E-04
Ac-228	0.00E+00	3.61E-01	8.95E-01
Ag-108	0.00E+00	6.10E-01	1.62E+00
Ag-108m	0.00E+00	0.00E+00	1.62E+00
Ag-109m	0.00E+00	0.00E+00	3.27E-03
Ag-110	0.00E+00	1.25E+00	2.77E+00
Ag-110m	0.00E+00	6.55E-02	2.74E+00
Am-241	5.48E+00	0.00E+00	2.23E-02
Am-242	1.83E-01	1.83E-01	4.45E-06
Am-243	5.26E+00	0.00E+00	5.17E-02
At-217	7.07E+00	0.00E+00	2.38E-04
Au-198	0.00E+00	4.94E-03	4.05E-01
Ba-133	0.00E+00	0.00E+00	4.02E-01
Ba-137m	0.00E+00	0.00E+00	5.96E-01
Ba-140	0.00E+00	2.89E-01	1.49E-01
Be-7	0.00E+00	0.00E+00	4.98E-02
Be-10	0.00E+00	2.03E-01	0.00E+00
Bi-210	0.00E+00	3.89E-01	0.00E+00
Bi-212	0.00E+00	2.21E+00	2.39E+00
Bi-213	0.00E+00	1.28E+00	1.41E+00
Bi-214	0.00E+00	6.44E-02	1.48E+00
Br-82	0.00E+00	3.86E-01	3.02E+00
C-14	0.00E+00	4.95E-02	0.00E+00
Ca-45	0.00E+00	7.72E-02	0.00E+00
Cd-104	0.00E+00	4.51E-07	2.29E-01
Cd-109	0.00E+00	0.00E+00	1.49E-02
Ce-139	0.00E+00	0.00E+00	1.33E-01
Ce-141	0.00E+00	1.45E-01	6.98E-02
Ce-144	0.00E+00	8.11E-02	1.57E-02

Table D3-3-1. (continued).

Radionuclide	ADE (MeV/dis)		
	Alpha	Beta	Gamma
Cf-252	5.93E+00	0.00E+00	2.04E-05
Cl-36	0.00E+00	2.49E-01	0.00E+00
Cm-242	6.10E+00	0.00E+00	2.37E-05
Cm-244	5.80E+00	0.00E+00	1.28E-05
Cm-248	4.65E+00	0.00E+00	8.80E-06
Co-57	0.00E+00	0.00E+00	1.20E-01
Co-58	0.00E+00	3.00E-02	8.06E-01
Co-60	0.00E+00	9.57E-02	2.50E+00
Cr-51	0.00E+00	0.00E+00	3.14E-02
Cs-134	0.00E+00	1.57E-01	1.55E+00
Cs-136	0.00E+00	1.00E-01	2.14E+00
Cs-137	0.00E+00	1.71E-01	5.96E-01
Er-169	0.00E+00	9.94E-02	1.31E-05
Eu-152	0.00E+00	5.71E-01	1.30E+00
Eu-154	0.00E+00	2.33E-01	1.19E+00
Eu-155	0.00E+00	4.52E-02	4.95E-02
Fe-55	0.00E+00	0.00E+00	1.47E-03
Fe-59	0.00E+00	1.18E-01	1.19E+00
Fr-221	6.35E+00	0.00E+00	3.28E-02
Fr-223	0.00E+00	3.41E-01	5.04E-02
Gd-152	2.15E+00	0.00E+00	0.00E+00
Gd-153	0.00E+00	0.00E+00	5.54E-02
H-3	0.00E+00	5.68E-03	0.00E+00
Hf-175	0.00E+00	0.00E+00	3.11E-01
Hf-181	0.00E+00	1.19E-01	5.18E-01
Hg-208	0.00E+00	5.77E-02	2.16E-01
I-125	0.00E+00	0.00E+00	2.30E-03
I-129	0.00E+00	2.30E-03	2.98E-03
I-131	0.00E+00	1.82E-01	3.78E-01
I-132	0.00E+00	4.86E-01	2.29E+00
I-133	0.00E+00	4.06E-01	6.02E-01

Table D3-3-1. (continued).

Radionuclide	ADE (MeV/dis)		
	Alpha	Beta	Gamma
In-113m	0.00E+00	0.00E+00	2.60E-01
Ir-192	0.00E+00	1.73E-01	8.10E-01
Kr-85	0.00E+00	4.79E-01	1.57E-01
La-140	0.00E+00	5.46E-01	2.07E+00
Mn-53	0.00E+00	0.00E+00	1.31E-03
Mn-54	0.00E+00	0.00E+00	8.35E-01
Mn-56	0.00E+00	8.31E-01	1.63E+00
Mo-99	0.00E+00	3.85E-01	2.82E-01
Na-22	0.00E+00	1.94E-01	1.27E+00
Na-24	0.00E+00	5.54E-01	4.12E+00
Nb-93m	0.00E+00	0.00E+00	1.95E-03
Nb-94	0.00E+00	1.48E-01	1.58E+00
Nb-95	0.00E+00	4.33E-02	8.27E-01
Ni-59	0.00E+00	0.00E+00	2.37E-03
Ni-63	0.00E+00	1.71E-02	0.00E+00
Np-237	5.02E+00	0.00E+00	2.02E-02
Np-238	0.00E+00	1.15E-01	1.72E-01
Np-240m	0.00E+00	5.90E-01	3.34E-01
P-32	0.00E+00	6.98E-01	0.00E+00
Pa-231	4.12E+00	0.00E+00	2.98E-02
Pa-233	0.00E+00	6.83-E01	1.55E-01
Pa-234m	0.00E+00	8.20E-01	1.14E-02
Pb-210	0.00E+00	6.55E-03	1.88E-03
Pb-212	0.00E+00	9.95E-02	1.17E-01
Pb-214	0.00E+00	2.19E-01	2.29E-01
Pm-147	0.00E+00	6.20E-02	0.00E+00
Po-210	5.31E+00	0.00E+00	0.00E+00
Po-212	8.75E+00	0.00E+00	0.00E+00
Po-214	7.69E+00	0.00E+00	0.00E+00
Po-216	6.78E+00	0.00E+00	0.00E+00
Po-218	6.00E+00	0.00E+00	0.00E+00

Table D3-3-1. (continued).

Radionuclide	ADE (MeV/dis)		
	Alpha	Beta	Gamma
Pr-143	0.00E+00	3.15E-01	0.00E+00
Pr-144	0.00E+00	1.21E+00	1.03E-02
Pu-238	5.49E+00	0.00E+00	2.78e-05
Pu-239	5.15E+00	0.00E+00	5.66E-05
Pu-240	5.15E+00	0.00E+00	2.72E-05
Pu-241	0.00E+00	5.23E-03	0.00E+00
Pu-242	4.89E+00	0.00E+00	2.26E-05
Pu-244	4.59E+00	0.00E+00	1.09E-03
Ra-224	3.81E+00	0.00E+00	9.49E-03
Ra-225	0.00E+00	9.40E-02	1.16e-02
Ra-226	4.77E+00	0.00E+00	6.10E-03
Ra-228	0.00E+00	9.90E-03	0.00E+00
Rb-86	0.00E+00	6.68E-01	9.43E-02
Re-188	0.00E+00	7.64E-01	5.80E-02
Rh-103m	0.00E+00	0.00E+00	1.72E-03
Rh-106	0.00E+00	1.41E+00	1.82E-01
Rn-220	6.29E+00	0.00E+00	5.50E-04
Rn-222	5.49E+00	0.00E+00	4.10E-05
Ru-103	0.00E+00	7.26E-02	4.62E-01
Ru-106	0.00E+00	1.00E-02	0.00E+00
S-35	0.00E+00	4.88E-02	0.00E+00
Sb-124	0.00E+00	3.82E-01	1.79E+00
Sb-125	0.00E+00	8.63E-02	4.14E-01
Sc-44	0.00E+00	5.97E-01	1.17E+00
Sc-46	0.00E+00	1.12E-01	2.01E+00
Se-75	0.00E+00	0.00E+00	3.86E-01
Sm-147	2.25E+00	0.00E+00	0.00E+00
Sn-113	0.00E+00	0.00E+00	4.29E-03
Sn-117m	0.00E+00	0.00E+00	1.58E-01
Sn-119m	0.00E+00	0.00E+00	3.86E-03
Sr-85	0.00E+00	0.00E+00	7.26E-01

Table D3-3-1. (continued).

Radionuclide	ADE (MeV/dis)		
	Alpha	Beta	Gamma
Sr-89	0.00E+00	5.83E-01	1.82E-04
Sr-90	0.00E+00	5.83E-01	0.00E+00
Sr-91	0.00E+00	6.54E-01	6.97E-01
Sr-92	0.00E+00	2.00E-01	1.34E+00
Ta-182	0.00E+00	1.23E-01	1.28E+00
Tc-99	0.00E+00	8.46E-02	1.25E-01
Tc-99m	0.00E+00	0.00E+00	1.27E-01
Te-125m	0.00E+00	0.00E+00	3.50E-02
Te-132	0.00E+00	5.94E-02	2.31E-01
Th-228	5.40E+00	0.00E+00	1.96E-03
Th-229	2.71E+00	0.00E+00	4.12E-02
Th-230	4.66E+00	0.00E+00	3.80E-04
Th-231	0.00E+00	7.84E-02	1.81E-02
Th-232	4.00E+00	0.00E+00	1.63E-04
Th-234	0.00E+00	4.45E-02	8.06E-03
Tl-204	0.00E+00	2.38E-01	0.00E+00
Tm-170	0.00E+00	3.15E-01	2.75E-03
U-232	6.32E+00	0.00E+00	2.43E-04
U-233	4.81E+00	0.00E+00	2.88E-04
U-234	4.76E+00	0.00E+00	1.49E-04
U-235	4.28E+00	0.00E+00	1.36E-01
U-236	4.49E+00	0.00E+00	4.08E-05
U-238	4.20E+00	0.00E+00	3.47E-05
U-240	0.00E+00	1.25E-01	6.72E-03
V-48	0.00E+00	0.00E+00	2.91E+00
W-185	0.00E+00	1.27E-01	2.51E-05
Xe-131m	0.00E+00	0.00E+00	2.01E-02
Xe-133	0.00E+00	1.00E-01	4.53E-02
Y-88	0.00E+00	7.71E-04	2.68E+00
Y-90	0.00E+00	9.31E-01	6.31E-01
Y-91	0.00E+00	6.02E-01	5.34E-01
Y-92	0.00E+00	1.44E+00	2.52E-01

Table D3-3-1. (continued).

Radionuclide	ADE (MeV/dis)		
	Alpha	Beta	Gamma
Y-93	0.00E+00	1.17E+00	8.91E-02
Yb-164	0.00E+00	0.00E+00	2.48E+00
Zn-65	0.00E+00	2.03E-03	5.66E-01
Zr-93	0.00E+00	1.96E-02	0.00E+00
Zr-95	0.00E+00	1.16E-01	8.00E-01

a. Data obtained from Raddecay software program.

Appendix D3
Attachment 2

Bioaccumulation Factors

CONTENTS

TABLES

Appendix D3
Attachment 2

Bioaccumulation Factors

D3-2-1. INTRODUCTION

The purpose of this section is to document and summarize the bioaccumulation factors (BAFs) developed for the ecological risk assessment (ERA) at the INEEL. Both organic and inorganic contaminants have been identified on the INEEL. The approach for developing BAFs for each class of contaminants and for different trophic levels at the INEEL will be presented in this document.

Transfer factors in animals are dependent on many factors including the nature and extent of the contaminant, the contaminant or radionuclide species, the animal species, the soil/chemical environment, a number of soil- and organism-related variables, as well as other conditions. Biological factors that may influence contaminant uptake and retention include species, sex, age, diet, tissue type, and season. Several of these factors are obviously highly interrelated, for example, diet and season.

The IAEA (1994) in the *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments* lists some of the difficulties in using transfer coefficients for radionuclides. Although directed at human health, these issues are also of concern for ecological receptors:

- The need for equilibrium
- Metabolic homeostasis
- Effects of chemical and physical form of radionuclide, and diet composition
- Influence of age

Also of concern are species differences and associated contaminates in the soil.

D3-2-1.1 The Need for Equilibrium

With a few exceptions, such as ^{131}I, most radionuclides will not have equilibrated in animal products before slaughter. Equally, few experiments are conducted for a sufficient length of time to enable equilibrium to be established. Hence, transfer coefficients derived from comparatively short-term experiments will underestimate equilibrium transfer coefficients. In some cases (for example, plutonium and americium) F_f values (transfer coefficient of meat) have been used for shorter for time periods, as the lifetime of the animal is too short for the radionuclide ever to reach equilibrium.

D3-2-1.2 Metabolic Homeostasis

Some elements (for example, sodium, magnesium, phosphorus, potassium and calcium), and therefore their radioisotopes, are subject to homeostatic control; hence, an increase in feed concentrations will not necessarily be reflected in tissues.

It is apparent that some essential elements do not bioaccumulate in biotic tissues in proportion to environmental levels. A lower transfer potential for essential trace elements (e.g., copper, zinc, chromium) may be due to their nutritional roles and effective homeostatic regulation in biological systems. Thus, essential elements are believed to be regulated within narrow ranges for each species, while tissue concentrations of nonessential elements or organics may be more dependent on ambient concentrations.

D3-2-1.3 Effects of Chemical and Physical Form of Radionuclide, and Diet Composition

The availability for gut uptake of radionuclides differs markedly, depending on the chemical and physical form, and on the constituents of the diet. Recent data obtained after the Chernobyl accident suggest that transfer coefficients for radiocesium in the first period after deposition were lower than those obtained once radiocesium was incorporated into plant tissue.

D3-2-1.4 Influence of Age

Young animals often have higher transfer coefficients than adults. Few transfer coefficient data that take this into account are available.

D3-2-1.5 Species Differences

Also of concern for ecological receptors are species differences. Diets containing contamination may vary in content and bioavailability to consumers. Even organisms exploiting similar food sources may show considerable differences in contaminant concentrations due to species- and metal-specific differences in the kinetics of assimilation and excretion (e.g., Janssen et al. 1991; Hopkin 1990). Carnivorous species may consume the whole animal, resulting in a different level of exposure than if solely the meat tissue is consumed. For example Sr-90 accumulation in the bones of antelope at the INEEL has been documented. Higher tropic-level species feeding on this species may or may not consume the bones resulting in different exposure (Markam, Halford, and Autenrieth 1980).

Although many metals accumulate, particularly in kidneys (also in livers) by being bonded to metallothioneins (e.g., Cd, Cu, Hg, Ni and Zn) (Elinder and Nordberg 1985), these organs are small and make up a minor part of carnivorous species food. Consequently, there may be no accumulation of these metals upwards in the foodweb. In terrestrial ecosystems we may therefore find the highest concentrations of these metals in the herbivorous species, e.g., at the bottom of the food-chain (Kalas et al. 1995).

Laskowski (1991) reviewed the existing knowledge concerning biomagnification and found that in some studies, interspecific differences in heavy metal concentrations within a trophic level have been found to be greater than the differences between trophic levels. Generally, predators consume all parts of their prey and considering only the meat will most likely highly underestimate the potential exposure.

D3-2-1.6 Associated Contaminants

A recent study has shown that the concentrations of other associated contaminants may impact the uptake of other metals in the soil. Laurinolli and Bendell-Young (1996) speculated that the enhanced accumulation of cadmium and zinc in liver of mice from a copper contaminated site (abandoned copper mine) was due to stimulation by the induction of metallothionein—a metal-binding protein in mammalian liver and kidney which sequesters and detoxifies some metals such as cadmium.

D3-2-1.7 Summary

In summary, appropriate and applicable studies on biotransfer in native species are not generally available. It has been shown by Nellessen and Fletcher (1993) that based on the contents of the UTAB—a reflection of the published literature—that there is essentially no plant-food-chain data available for approximately 75% of the hazardous substances monitored by the EPA. This is with the exception of pesticides for which there is a substantial amount of published information.

To fully assess the transfer potential of contaminants in terrestrial ecosystems, it would be necessary to monitor various physical, chemical, and biological compartments of the ecosystem; measurements of contaminant levels in soil alone may not be an adequate indicator of biotransfer potential. Limited site-specific data are available for this effort and the INEEL risk assessment relies heavily on the available literature. As a result, the degree of applicability of literature biotransfer factors to organisms at INEEL is uncertain.

D3-2-2. DEVELOPMENT OF TRANSFER FACTORS FOR USE IN ERA

Attachment 1 to this appendix contains the final TF used to develop screening level valves and to perform the risk assessment calculations. The following sections discuss their determination.

D3-2-2.1 Inorganics

Literature supports the development of transfer factors (TFs) for meat and dairy products (primarily herbivores) reflecting an emphasis placed on human health. For inorganic contaminants, this information is summarized in several sources including; IAEA (1994), Ng et al. (1979), EPA (1989), and National Council of Radiation Protection and Measurements (NCRP) (1984). These were evaluated as well as any additional studies that were available and the most applicable BAFs was selected. As shown in Table D3-2-1, the overall summaries and the values provided by comprehensive BAFs were given some preference in the selection process since these studies are well documented and accepted by decision-makers.

D3-2-2.2 Organics

The TFs for organics were calculated using the following allometric equation presented in Travis and Arms (1988):

$\log TFs = -7.6 + \log K_{ow}$.

Log partitioning coefficients (K_{ow}s) were taken from Montgomery and Welkom (1990).

Table D3-2-1. Transfer factors (feed to meat) for inorganic contaminants at the INEEL.

	Baes et al. (1984)	EPA (1989)	NCRP (1984)	Ng et al. (1979)	IAEA (1994)	Other Sources	Suggested for INEEL
Aluminum (Al)	1.5E-03	Not given	Not given	Not given			1.5E-03
Americium (Am)	3.5E-06	5.5E-3	2.0E-04	3.6E-06	4.1E-03 (lamb) 6E-03 (poultry)		5.5E-03
Antimony (Sb)	1.0E-03	2.0E-01	4.0E-03	1.2E-03	4E-05 (beef)	See section 3-2-3.1	
Arsenic (As)	2.0E-03	4.0E-02	Not given	Not given		See section 3-2-3.1	
Barium (Ba)	1.5E-04	1.5E-01	3.2E-03	1.6E-04	2E-04 (beef) 9E-03 (poultry)	Hope & Miller (1996) found that Ba bioaccumulation factors are low for all terrestrial receptors examined and the concentrations decrease in increasing trophic levels. Johnson et al., 1988 2.3E-04 (cow), 1.3E-05 (goat)	1.5E-01
Beryllium (Be)	1.0E-03	1.0E-02	Not given	Not given			1.0E-02
Cadmium (Cd)	5.5E-04	5.5E-01	Not given	Not given	1.5E-02 (pork) 8E-01 (poultry)	Laurinolli & Bendel-Young (concentrations in liver) 0.47 (reference area) to 0.27 (Cu mine site). This is a temperate site. See section 3-2-3.1	
Calcium (Ca)	7.0E-04	3.5E+00	Not given	Not given	2E-03 (beef) 4E-02 (poultry)		3.5E+00
Carbon (C)	not given	0.0	Not given	Not given			0.0E+00
Cerium (Ce)	7.5E-04	1.0E-02	1.2E-03	7.5E-04	2E-04 (lamb) 4E-03 (poultry)	Transfer factors can be calculated for mono and ruminants by body wt. (Ng et al., 1979)	1.0E-02
Cesium (Cs)	2.0E-02	8.0E-02	4.0E-03	2.0E-02	8E-01 (reindeer-winter) 1E+01 (poultry)		8.0E-01
Chloride (Cl)	8.0E-02	Not given	Not given	Not given	2E-02 (beef)		8.0E-02
Chromium (Cr)	5.5E-03	7.5E-03	2.4E-03	5.6E-03	9E-03 (beef)	See section 3-2-3.1	7.5E-03
Cobalt (Co)	2.0E-02	2.0E-02	1.3E-02	2.1E-02	6.2E-02 (sheep) 2 (poultry)		2.0E-02
Copper (Cu)	1.0E-02	4.0E-01	8.0E-03	1.1E-02	3.9E-02 (sheep) 5E-01 (poultry)	Laurinolli & Bendel-Young (1996) (concentrations in liver) 0.53 (reference area) to 0.12 (Cu mine site). This is a temperate site. See section 3-2-3.1	
Curium (Cm)	3.5E-06	8.5E-04					8.5E-04
Europium (Eu)	5.0E-03	1.0E-02					1.0E-02
Hydrogen (H)	not given	0.0			^{3}H (OBT) 2.8E-02 (goat)		2.8E-02

Table D3-2-1. (continued).

	Baes et al. (1984)	EPA (1989)	NCRP (1984)	Ng et al. (1979)	IAEA (1994)	Other Sources	Suggested for INEEL
Iodine (I)	7.0E-03	1.0E+00	2.9E-03	7.2E-03	4E-02 (beef) 1E-02 (poultry)		1.00E+00
Iron (Fe)	2.00E-02	4.00E-03	4.0E-02	1.7E-02	2.6E-02 (pork) 1 (poultry)		2.00E-02
Lanthanum (La)	3.0E-04	not given	2.0E-04	2.9E-04	1E-01 (poultry)		3.0E-04
Lead (Pb)	3.0E-04	4.5E-02			4E-04 (beef)	Pb-210 beef fraction element ingested daily in kg of flesh 2E-04 to 2E-03 (Chester and Garten 1980). See section 3-2-3.1.	
Manganese (Mn)	4.0E-04	2.5E-01	8.0E-04	3.9E-04	5.9E-03 (sheep) 5E-02 (poultry)		2.5E-01
Mercury (Hg)	2.5E-01	9.0E-01			3E-02 (poultry)	See section 3-2-3.1	
Molybdenum (Mo)	6.0E-03	2.5E-01	8.0E-03	6.1E-03	1E-03 (beef) 1 (poultry)	Johnson et al, 1988 1.0E-03 (cow)	2.5E-01
Neodymium (Nd)	3.0E-04	1.0E-02	3.3E-03	2.9E-04	9E-02 (poultry)		1.0E-02
Niobium (Nb)	2.5E-01	2.0E-02	2.8E-01	2.5E-01	3E-04 (sheep) 3E-04 (poultry)	Johnson et al., 1988, 2.6E-07 (cow); 6.0E-05 (goat)	2.5E-01
Nickel (Ni)	6.0E-03	6.0E-02	5.3E-02	5.9E-03	5E-03 (beef)		6.0E-02
Neptunium (Np)	5.5E-05	1.0E-01			1E-03 (beef)		1.0E-01
Phosphorus (P)	5.5E-02		4.6E-02	5.5E-02	5E-02 (beef)		5.5E-02

Table D3-2-1. (continued).

	Baes et al. (1984)	EPA (1989)	NCRP (1984)	Ng et al. (1979)	IAEA (1994)	Other Sources	Suggested for INEEL
Plutonium (Pu)	5.0E-07	4.5E-04	1.4E-05	1.0E-06	3.1E-03 (lamb) 3E-03 (poultry)	1.0E-04 to 1.0E-01 (Hakonson, 1975) 1E-02 (mL) there was an apparent increase in the biological availability of Pu-238 relative to that of Pu-239/40 in the environment 1.9E-03 to 6.8E-04 Romeny et al., 1970, sampled different jackrabbits and kangaroo rats tissues at the Nevada Test Site. They found that there was considerable variation in tissue burdens. They also found that the accumulation of residual Pu-239 was highest in bone tissue; considerable amounts also were found in lung tissues. High levels were also found in the gastrointestinal tracts indicating the ingestion is also an important route through which these small mammals maintained contact with Pu in the environment.	4.5E-04
Polonium (Po)	9.5E-05	2.5E-02			5E-03 (beef)		2.5E-02
Praseodymium (Pr)	3.0E-04	1.0E-02	4.7E-03	2.9E-04	3E-02 (poultry)		1.0E-02
Radium (Ra)	2.5E-04	1.5E-02			9E-04 (beef)	Ra-226 beef fraction element ingested daily in kg of flesh undetectable to 2E-03 (Hakonson, 1975-review)	1.5E-02
Rhodium (Rh)	2.0E-03	1.5E-01	1.5E-03	2.0E-03			1.5E-01
Rubidium (Rb)	1.5E-02	1.5E-01	3.1E-02	1.4E-02	6.9E-01 (lamb)		1.5E-01
Ruthenium (Ru)	2.0E-03	7.5E-02	4.0E-01	2.0E-03	1.5E+00 (lamb) 8.0E+00 (poultry)		1.5E+00
Selenium (Se)	1.5E-02	2.5E-02			3.2E-01 (pork) 9 (poultry)		3.2E-01
Silver (Ag)	3.0E-03	4.0E-01	1.7E-02	2.9E-03	2E-02 (pork) 2.0E+00 (poultry)		4.0E-01
Sodium (Na)	5.5E-02	7.5E-02	3.0E-02	5.4E-02	8E-02 (beef)		7.5E-02
Strontium (Sr)	3.0E-04	2.5E+00	6.0E-04	3.0E-04	3.3E-01 (lamb) 8E-02 (poultry)	See section 3-2-3.1	
Technetium (Tc)	8.5E-03	9.5E+00	4.0E-01	8.7E-03	2.2E-04	Johnson et al., 1988, 7.3E-07 (cow), 2.2E-04 (goat)	9.5E+00

Table D3-2-1. (continued).

	Baes et al. (1984)	EPA (1989)	NCRP (1984)	Ng et al. (1979)	IAEA (1994)	Other Sources	Suggested for INEEL
					(Tc-99m goat) 3E-02 (poultry)		
Tellurium (Te)	1.5E-02	2.5E-02	7.7E-02	1.5E-02	7E-03 (beef) 6E-01 (poultry)	Johnson et al., 1988, 7.0E-03 (cow), 2.4E-03 (goat)	2.5E-02
Tin (Sn)	8.0E-02	3.0E-02	Not given	Not given			8.0E-02
Thallium (Tl)	4.0E-02	4.0E-03	Not given	Not given			4.0E-02
Thorium (Th)	6.0E-06	8.5E-04	Not given	Not given			8.5E-04
Tungsten (W)	4.5E-02	4.5E-02	1.3E-03	4.7E-02	4E-02 (beef)		4.5E-02
Uranium (U)	2.0E-04	8.5E-03	Not given	Not given	6.2E-02 (pork) 1.0E+00 (poultry)		6.2E-02
Vanadium (V)	2.5E-03	Not given	Not given	Not given			2.5E-03
Yttrium (Y)	3.0E-04	1.5E-02	4.6E-03	2.9E-04	1E-03 (beef) 1E-02 (poultry)		1.5E-02
Zinc (Zn)	1.0E-01	1.5E+00	Not given	Not given	4.1 (sheep) 7.0E+00 (poultry)	Laurinolli & Bendel-Young (1996) (concentrations in liver) 1.16 (reference area) to 0.66 (Cu mine site). This is a temperate site. See section 3-2-3.1	
Zirconium (Cr)	5.5E-03	2.0E-03	3.4E-02	4.6E-03	2E-05 (goat) 6E-05 (poultry)	Johnson et al., 1988, 1.2E-06 (cow), 2.0E-05 (goat)	5.5E-03

TFs outside the range of the Travis and Arms (1988) study were assigned values at the limits of the evaluation. Table D3-2-2 presents the values for organics identified as present on the INEEL as calculated using the allometric equation. This equation presents the tissue concentration.

D3-2-3. ADDITIONAL DEVELOPMENT OF SELECTED INORGANICS TRANSFER FACTORS

Some effort was made to evaluate biotransfer of selected metals in different trophic levels. This effort resulted in allowing more specific information to be incorporated into the risk assessment. The general pattern of metals accumulation in soil invertebrates is toward higher concentrations in spiders (*Arachnida*) and detritivores than in herbivorous and carnivorous species (Stafford 1988; Ainsworth 1990a). Because earthworms are an important link in the food chains of insectivorous and carnivorous animals, their uptake of soil-associated chemicals has been more extensively studied than that of other terrestrial soil-dwelling invertebrates. Earthworms at INEEL occur only on irrigated lawns but may be

Table D-3-2-2. Transfer factors (feed to meat) used in ecological risk assessments at the INEEL for organic contaminants.

CAS #	Contaminant	K_{ow}	Calculated BAF
75-34-4	1,1-Dichloroethylene	6.17E+01	1.55E-06
71-55-6	1,1,1 Trichloroethane	3.16E+02	7.94E-06
76-13-1	1,1,2-Trichloro-1,2,2-Trifluoroethane	1.00E+02	2.51E-06
120-82-1	1,2,4-Trichlorobenzene	2.00E+04	5.02E-04
79-34-5	1,1,2,2-Tetrachloroethane	2.45E+02	6.15E-06
106-46-7	1,4-Dichlorobenzene	3.98E+03	1.00E-04
78-93-3	2-Butanone	1.82E+00	4.57E-08
108-41-8	2-Chlorotoluene	1.90E+03	4.77E-05
591-78-6	2-Hexanone	NA	NA[a]
91-57-6	2-Methylnaphthalene	1.20E+04	3.01E-04
88-75-5	2-Nitrophenol	5.75E+01	1.44E-06
67-63-0	2-Propanol	NA	NA
51207-31-9	2,3,7,8,-Tetrachloro dibenzodioxin	NA	NA
105-67-9	2,4- Dimethylphenol	NA	NA
94-75-7	2,4-Dichlorophenoxyacetic acid	6.46E+02	1.62E-05
121-14-2	2,4-Dinitrotoluene	1.00E+02	2.51E-06
106-47-8	4-Chloroaniline	6.76E+01	1.70E-06
106-44-5	4-Methylphenol	9.33E+01	2.34E-06
59-50-7	4-Chloro-3-methylphenol (CMP)	NA	NA
83-32-9	Acenaphthene	1.00E+04	2.51E-04

Table D3-2-2. (continued).

CAS #	Contaminant	K_{ow}	Calculated BAF
67-64-1	Acetone	5.75E-01	1.44E-08
75-05-8	Acetonitrile	NA	NA
107-13-1	Acrylonitrile	NA	NA
120-12-7	Anthracene	2.82E+04	7.08E-04
71-43-2	Benzene	1.32E+02	3.32E-06
8032-32-4	Benzine	NA	NA
56-55-3	Benzo(a)anthracene	3.98E+05	1.00E-02
50-32-8	Benzo(a)pyrene	1.15E+06	2.89E-02
205-99-2	Benzo(b)fluoranthene (BbF)	1.15E+06	2.89E-02
207-08-9	Benzo(k)fluoranthene	1.15E+06	2.89E-02
191-24-2	Benzo(g,h,i)perylene	3.24E+06	8.14E-02
71-36-3	Butyl alcohol	NA	NA
85-68-7	Butylbenzylphthalate (BBP)	6.31E+04	1.59E-03
75-15-0	Carbon disulfide	1.00E+02	2.51E-06
56-23-5	Carbon tetrachloride	6.76E+02	1.70E-05
67-66-3	Chloroform	9.33E+01	2.34E-06
74-87-3	Chloromethane (Methyl chloride)	9.50E-01	2.39E-08
218-01-9	Chrysene	4.07E+05	1.02E-02
57-12-5	Cyanide	NA	NA
112-31-2	Decanal	NA	NA
132-64-9	Dibenzofuran	NA	NA
75-71-8	Dichlorodifluoromethane	1.45E+02	3.64E-06
117-81-7	Di-2-ethylhexyl-phthalate (DEHP)	9.50E+03	2.39E-04
84-66-2	Diethyl phthalate	3.16E+02	7.94E-06
131-11-3	Dimethyl phthalate	1.32E+02	3.32E-06
84-74-2	Di-n-butylphthalate	3.98E+05	1.00E-02
1746-01-6	Dioxin	5.25E+06	1.32E-01
117-84-0	Di-n-octylphthalate	1.58E+09	3.97E+01
64-17-5	Ethanol (Ethyl alcohol)	4.79E-01	1.20E-08
100-41-4	Ethylbenzene	1.41E+03	3.54E-05
206-44-0	Fluoranthene	7.94E+04	1.99E-03
86-73-7	Fluorene	1.58E+04	3.97E-04
50-00-0	Formaldehyde	1.00E+00	2.51E-08

Table D3-2-2. (continued).

CAS #	Contaminant	K_{ow}	Calculated BAF
302-01-2	Hydrazine	8.32E-04	2.09E-11
78-59-1	Isophorone	5.01E+01	1.26E-06
193-39-5	Indeno(1,2,3)pyrene	3.16E+06	7.94E-02
7439-97-6	Mercury (Organic)	NA	NA
67-56-1	Methanol (Methyl alcohol)	NA	NA
108-10-1	Methyl isobutyl ketone	NA	NA
75-09-2	Methylene chloride	2.00E+01	5.02E-07
103-65-1	n-Propylbenzene	NA	NA
91-20-3	Naphthalene	2.76E+03	6.93E-05
78-48-8	Orthophosphate	NA	NA
11097-69-1	PCBs–Aroclor 1254	1.07E+06	2.69E-02
11096-82-5	PCBs–Aroclor 1260	1.38E+07	3.47E-01
1336-36-3	PCBs	1.10E+06	2.76E-02
85-01-8	Phenanthrene	2.88E+04	7.23E-04
108-95-2	Phenol	2.88E+01	7.23E-07
107-12-0	Propionitrile	NA	NA
129-00-0	Pyrene	7.59E+04	1.91E-03
143-33-9	Sodium cyanide	NA	NA
18496-25-8	Sulfide	NA	NA
7664-93-9	Sulfuric acid	NA	NA
26140-60-3	Terphenyl	NA	NA
127-18-4	Tetrachloroethylene	3.98E+02	1.00E-05
109-99-9	Tetrahydrofuran	NA	NA
108-88-3	Toluene	5.37E+02	1.35E-05
	Total Petroleum Hydrocarbon	NA	NA
126-73-8	Tributyl phosphate	NA	NA
79-01-6	Trichloroethylene (Trichloroethene)	2.40E+02	6.03E-06
15625-89-5	Trimethylopropane-triester	NA	NA
108-05-4	Vinyl acetate	NA	NA
1330-20-7	Xylene (mixed)	1.83E+03	4.60E-05

a. NA—could not calculate-use a default of 1.0.

used as an example of invertebrate bioaccumulation. In general, earthworms may provide a good indication of the "worst case" of metal uptake by soil-dwelling invertebrates (Stafford 1988). Thus, BAFs for earthworms may be regarded as a conservative surrogate for other invertebrates. Further, accumulation of certain metals in insectivorous mammals reflects their bioavailability to earthworms (Ma 1987; Scanlon 1987; Hegstrom and West 1989).

The relatively well-studied earthworm system demonstrates some of the complexities of predicting the biotransfer of metals in terrestrial ecosystems. The body concentration of a metal in earthworms is determined by its concentration in soil, the intrinsic rate of bioaccumulation, and the tolerance of the organism to the element. It also depends on the influence of several edaphic factors, notably soil pH, organic matter content, calcium content, and cation exchange capacity (CEC) (Ma 1982; Ma et al. 1983; Corp and Morgan 1991). The bioavailability of several metals to worms appears to be greater in sandy than loamy soils (Ma 1982).

CEC, the total amount of cations exchangeably adsorbed by the soil exchange complex, provides an estimate of the capacity of the soil to adsorb heavy metals and gives a measure of the ability of soils to retain these metals against uptake by earthworms (Ma 1982). Significant negative correlations were found between the concentration factor (CF) and the pH of the soil for several metals, including zinc (Ma 1982). For copper, a negative correlation was found with soil organic matter (Ma 1982). Further, the presence and concentration of other metals can have a significant effect on worm uptake of particular metals (Back 1990).

In view of the many gaps in our knowledge of metal biotransfer in the terrestrial environment, the BAFs for the metals in the following subsection are highly uncertain. An effort has been made to select factors that are protective for use in the assessments at INEEL. All of these values are in terms of dry weight. The results of this effort are provided in Table D3-2-3.

D3-2-3.1 Selected Metals Analysis

D3-2-3.1.3 Antimony

The biotransfer of antimony within food chains in a grassland ecosystem in the vicinity of an antimony smelter was studied by Ainsworth (1990a,b 1991). Several mammalian and macroinvertebrate species at different trophic levels, as well as food plants, were examined in areas with soil concentrations of antimony ranging from 6.9 mg/kg (Ainsworth 1990b) to 386 mg/kg near the smelter. Tissue concentrations in all species examined were low relative to both soil and dietary concentrations, indicating that for antimony bioaccumulation in potential terrestrial food chains is low.

The general trend for invertebrates was toward higher concentrations in the detritivores (oligochaetes, diplopods, isopods, and dipteran larvae) than in the herbivorous and predatory groups (e.g., lepidopterans and staphylinids). This trend indicates a pattern of food chain biominification for this metal. As shown in Table D3-2-4, mean BAFs ranged from 0.04 in lepidopterans to 0.9 in oligochaeta (Ainsworth 1990b), with a geometric mean for all macroinvertebrates of 0.1.

Two herbivorous species (the rabbit [*Oryctolagus cuniculus*] and the short-tailed field vole [*Microtus agrestis*] and one insectivorous species (the common shrew [*Sorex araneus*]) of mammals were examined as available at the study locations. Antimony concentrations were measured in individual organs rather than the whole body, limiting the usefulness of these data for purposes of estimating food chain exposure. To ensure that bioaccumulation is not underestimated, BAFs were calculated with data from the liver, which contained the highest concentrations of antimony in all species. Results are

Table D3-2-3. PUFs and BAFs (or CFs) for selected INEEL inorganic contaminants[a] (unitless).

Contaminants	PUF[b]	BAF[c] for Insectivores	BAF for Carnivores[e]	BAF for Omnivores[f]
Antimony	2.0E-01	9.0E-01	5.5E-03	9.0E-01
Arsenic	4.0E-02	1.0E+00	4.0E-02	1.0E+00
Cadmium	5.5E-01	1.1E+00	1.9E+00	1.9E+00
Chromium	1.9E-01	6.0E-02	2.0E-01	2.0E-01
Copper	4.0E-01	1.0E+00	2.0E-01	1.0E+00
Lead	4.5E-02	3.0E-01	6.0E-01	6.0E-01
Mercury	9.0E-01	4.0E-01	7.0E-01	7.0E-01
Strontium[g]	7.5E-02	1.5E+00	1.5E+00	1.5E+00
Zinc	1.5E+00	1.0E+00	7.0E-01	1.0E+00

a. Values and or literature for inorganics come from Baes et al., (1984).

b. PUF = Plant uptake factor, appropriate for use with AV100 and M100 Functional Groups.

c. Bioaccumulation factor.

d. BAFs or CFs for insectivores, appropriate for AV200 and M200 Functional Groups.

e. BAFs or CFs for carnivorous, appropriate for AV300 and M300 Functional Groups.

f. BAFs or CFs for omnivores, appropriate for AV400 and M400 Functional Groups.

g. Site-specific data (VanHorn et al., 1995).

Table D3-2-4. Mean BAFs for antimony in terrestrial macroinvertebrates.[a]

Taxonomic group	Mean BAF (± SD)
Isopoda	0.13 ± 0.13
Diplopoda	0.13 ± 0.12
Lepidoptera	0.04 ± 0.02
Diptera	0.20 ± 0.07
Coleoptera	0.08 ± 0.05
Lycosidae	0.08 ± 0.05
Oligochaeta	0.89 ± 0.21
Overall geometric mean	0.14

a. Data from Ainsworth (1990a).

shown in Table D3-2-5. Although these BAFs are clearly overestimated because antimony concentrations in liver are undoubtedly higher than whole-body concentrations, they are still considerably less than unity,

indicating no biomagnification of antimony in small mammals. However, the insectivorous shrew appeared to accumulate more antimony than the herbivorous species, perhaps due to greater bioavailability of invertebrate-borne metal (Ainsworth 1990b). The geometric mean BAF for the three species was approximately 0.002. A BAF of 1.0 was used for all functional groups to be protective at the screening level.

D3-2-3.1.4 Cadmium

Large differences in cadmium concentrations among arthropod and mammalian species collected at the same site have been observed. Laskowski (1991) summarized available data on cadmium bioaccumulation in terrestrial food chains. Organisms considered included macroinvertebrates and the carnivorous shrew (*S. araneus)*, and encompassed four trophic levels: herbivores, carnivores, top carnivores, and detritivores. Of 37 reported tissue:dietary concentration ratios identified in the literature for cadmium, 26 were greater than 1.0 (Laskowski 1991). Geometric mean values for herbivorous, carnivorous, and detritivorous invertebrates were 1.1, 1.5, and 2.4, respectively. The mean tissue:diet ratio for the shrew was 1.7 (Laskowski 1991). However, the slope of the regression line of dietary to tissue concentrations for all species was only slightly greater than 1.0 (1.3), indicating little potential for biomagnification in the terrestrial food chain. These data, summarized in Table D3-2-6 were used to estimate the following BAFs for terrestrial organisms.

Assuming a plant uptake factor (PUF) of 0.55 for cadmium (Baes et al. 1984), a geometric mean tissue-to-soil BAF ratio of 0.6 can be estimated for herbivorous invertebrates by multiplying the two factors:

$$\textit{Herbivorous Invertebrate } BAF_{cadmium} = PUF_{cadmium} \times \frac{[Cadmium]\ in\ invertebrate}{[Cadmium]\ in\ plants} \qquad \text{(D3-2-1)}$$

This value is in good agreement with BAFs for other herbivorous invertebrates reported subsequently (e.g., Lindqvist 1992; Janssen and Hogervorst 1993).

BAFs for cadmium in earthworms and other detritivores are typically higher than those for other soil macroinvertebrates. Uptake by earthworms has been shown to be dependent on many soil parameters, especially pH (Ma 1982), as well as the presence of other metals in the soil (Beyer et al. 1982). Data for earthworms were reviewed by Romijn et al. (1991), who observed that the BAF is not

Table D3-2-5. BAFs for antimony in small mammals.[a]

Taxonomic group	Mean BAF
Short-tailed field vole	7.8×10^{-4}
Rabbit	3.4×10^{-3}
Common shrew	6.0×10^{-3}
Overall mean	2.5×10^{-3}

a. Data from Ainsworth (1990a).

Table D3-2-6. Summary of cadmium uptake factors and estimated BAFs in terrestrial ecosystems.

Taxonomic Group	Geometric mean ratio of tissue:diet cadmium concentration (dry weight)[a]	BAF
Herbivorous invertebrate	1.1	0.6
Carnivorous invertebrate	1.5	0.9
Detritivorous invertebrate	2.4	7.1[b]
Small mammal (*S. araneus*)	1.7	1.9

a. Data from Laskowski (1991).

B. Derived from a regression equation (Ma 1983) as discussed in text.

constant but is inversely related to soil concentration. Thus, less cadmium is taken up relative to soil concentrations as concentrations increase. Ma (1982) defined the relationship between soil and worm concentrations of cadmium as:

$$ln([Cadmium]\ in\ worm\ tissue) = 5.538 + 0.664\ ln([Cadmium]\ in\ soil) - 0.40\ pH \quad \text{(D3-2-2)}$$

$$Earthworm\ BAF_{cadmium} = \frac{[Cadmium]\ in\ worm\ tissue}{[Cadmium]\ in\ soil} \quad \text{(D3-2-3)}$$

Given the pH ranges identified at the INEEL facility (Martin et al. 1992) and the concentrations of cadmium in the soil (2.2 mg/kg), the earthworm BAFs developed using this equation will range from approximately 4.5 to 7.0.

Assuming that carnivorous invertebrates consume primarily herbivorous species, a BAF of 0.9 can be estimated for carnivorous insects by multiplying the estimated BAF for these prey items (0.6) by the mean ratio of cadmium concentrations in carnivores and herbivores (1.5) reported by Laskowski (1991):

$$Carnivorous\ Invertebrate\ BAF_{cadmium} = BAF\ herbivores\ x \frac{[Cadmium]\ in\ carnivores}{[Cadmium]\ in\ prey} \quad \text{(D3-2-4)}$$

Interspecific variation in cadmium accumulation among mammalian species in the same environment has been observed in several studies (e.g., Anthony and Kozlowski 1982; Scanlon 1987). Data appear to be most abundant for the shrew, which also typically has higher tissue concentrations than herbivorous/omnivorous small mammals (Hunter et al. 1987). Assuming that the BAF of organisms consumed by shrews is approximately 1.1 (the geometric mean of values derived in the equation for carnivorous invertebrates), and the ratio of shrew body burden to prey body burden is 1.7 (Laskowski 1991), a shrew BAF can be calculated by multiplying these two factors:

$$Shrew\ BAF_{cadmium} = BAF\ prey\ x \frac{[Cadmium]\ in\ shrew}{[Cadmium]\ in\ prey} \quad \text{(D3-2-5)}$$

Thus, the BAF for cadmium in small mammals is conservatively estimated as 1.9.

D3-2-3.1.5 Chromium

Trivalent chromium is an essential trace element found in all living organisms. Chromium deficiency may result in irreversible metabolic damage. Several researchers have observed that chromium is biominified rather than biomagnified in terrestrial ecosystems. Indeed, in every example reported, chromium concentrations in animals were equal to or lower than those in soils and dietary items (reviewed by Outridge and Scheuhammer 1993). For example, chromium was the least accumulated of eight metals examined by Ma (1982) in earthworms, with a geometric mean of only 0.06 (chromium species not reported). In a recent study, BAFs for earthworms were observed to be concentration-dependent (Van Gestel et al. 1993). Further, Beyer et al. (1990) observed no relationship between chromium concentrations in soil and biota at disposal facilities for dredged material. The validity of BAFs derived in the absence of significant correlation is questionable. Such observations indicate that, as expected, chromium uptake is tightly regulated, and is unlikely to be significantly accumulated in the food chain.

In the absence of more definitive data, a BAF of 0.06 is recommended for invertebrates shown in Table D3-2-7. Because earthworms generally accumulate metals more avidly than other invertebrates, this value is likely to be conservative for soil-dwelling arthropods.

For small mammals, a BAF of 6×10^{-5} has been estimated (VanHorn et al. 1995) as the product of the assimilation efficiency of ingested hexavalent ^{51}Cr in cotton rats (0.008; Taylor and Parr 1978) and the PUF for chromium (0.0075; Baes et al. 1984). However, because assimilation efficiency refers to dose absorption (i.e., bioavailability) rather than bioaccumulation, this manipulation is inappropriate. The geometric mean BAF for chromium in the house mouse (*Mus musculus*) (0.2) determined by Beyer et al. (1990) is shown in Table D3-2-8.

D3-2-3.1.6 Copper

Laskowski (1991) summarized available data on copper bioaccumulation in terrestrial food chains. Organisms considered included macroinvertebrates and the carnivorous shrew (*S. araneus*), and encompassed four trophic levels: herbivores, carnivores, top carnivores, and detritivores. Of 37 reported tissue: dietary concentration ratios identified in the literature for copper, 22 were greater than 1.0 (Laskowsi 1991). Geometric mean values for herbivorous, carnivorous, and detritivorous invertebrates were 2.5, 1.1, and 0.3, respectively. The mean tissue: diet ratio for the shrew was 0.2 (Laskowski 1991). However, the slope of the regression line of dietary to tissue concentrations for all species was less than 1.0 (0.83), suggesting regulation of copper ion concentrations in terrestrial organisms. These data, summarized in Table D3-2-8, were used to estimate the following BAFs for terrestrial organisms.

Table D3-2-7. Geometric mean BAFs for chromium in terrestrial ecosystems.

Taxonomic Group	BAF
Earthworm, arthropod[a]	0.06
Small mammal (*Mus musculus*)[b]	0.20

a. Data from Ma (1982).

b. Data from Beyer et al. (1990).

Table D3-2-8. Summary of copper uptake factors and estimated BAFs in terrestrial ecosystems.

Taxonomic Group	Geometric Mean Ratio of Tissue: Diet Copper Concentration (dry weight)[a]	BAF
Herbivorous invertebrate	2.5	1.0
Carnivorous invertebrate	1.1	1.1
Detritivorous invertebrate	0.3	0.34[b]
Small mammal (*S. araneus*)	0.2	0.2

a. Data from Laskowski (1991).

b. Calculated using regression equation from Ma et al. (1983).

The bioavailability of copper to earthworms appears to be strongly influenced by copper concentration and soil type, but not by soil pH (Ma 1982; Ma et al. 1983; Corp and Morgan 1991). As for cadmium and other metals, less copper is taken up relative to soil concentrations as these concentrations increase. Ma et al. (1983) defined the relationship between tissue and soil concentrations of copper in soil (in mg/kg dry weight) near a zinc smelter as:

$$[Copper]\ in\ worm\ tissue = 14.88 + 0.344\ x\ [Copper]\ in\ soil \qquad \text{(D3-2-6)}$$

$$Earthworm\ BAF_{copper} = \frac{[Copper]\ in\ worm\ tissue}{[Copper]\ in\ soil} \qquad \text{(D3-2-7)}$$

showing the decreasing BAF with increasing soil concentration. Corp and Morgan (1991) observed a similar relationship in worms exposed to naturally metalliferous soils. In addition, concentration-dependence of the copper BAF for isopods was recently reported (Hopkin et al. 1993). This relationship can be used to calculate site-specific BAFs for copper in earthworms. This formula yields BAFs for earthworms of around 6 for soil concentrations of 1 to 10 mg/kg, 0.9 for 10 to 100 mg/kg, and 0.4 for 100 to 1,000 mg/kg.

Assuming a PUF of 0.4 for copper (Baes et al. 1984), a mean tissue:soil BAF of 1.0 can be estimated for herbivorous invertebrates by multiplying this factor by the ratio of copper in animal:plant tissues (2.5):

$$Herbivorous\ Invertebrate\ BAF_{copper} = PUF_{copper}\ x\ \frac{[Copper]\ in\ invertebrates}{[Copper]\ in\ plants} \qquad \text{(D3-2-8)}$$

This value is in good agreement with subsequently reported BAFs for copper in other herbivorous invertebrates (e.g., Lindqvist 1992; Janssen and Hogervorst 1993). Assuming that carnivorous invertebrates consume primarily herbivorous species, a BAF of 1.1 can be estimated for carnivorous insects by multiplying the estimated BAF for these prey items (1.0) by the geometric mean ratio of copper concentrations in carnivores and herbivores (1.1) reported by Laskowski (1991):

$$Carnivorous\ Invertebrate\ BAF_{copper} = BAF_{herbivores} \times \frac{[Copper]\ in\ carnivores}{[Copper]\ in\ prey} \quad \text{(D3-2-9)}$$

This value is somewhat higher than reported in other studies (e.g., Beyer et al. 1990; Janssen and Hogervorst 1993).

Assuming that the BAF of organisms consumed by shrews is approximately 1.1 (the geometric mean of values derived above for herbivorous and carnivorous macroinvertebrates), and the ratio of shrew body burden to prey body burden is 0.2 (Laskowski 1991), a shrew BAF can be calculated by multiplying these two factors:

$$Shrew\ BAF_{copper} = BAF_{prey} \times \frac{[Copper]\ in\ shrew}{[Copper]\ in\ prey} \quad \text{(D3-2-10)}$$

Thus, the BAF for copper in shrews is around 0.2 as listed in Table D3-2-8. This BAF agrees with a BAF value estimated for house mice by Beyer et al. (1990).

D3-2-3.1.7 Lead

Soil pH and CEC are prime factors in predicting the uptake and accumulation of lead in earthworms (e.g., Ma 1982). Organic matter, calcium, and the presence of other metals are also influential (Terhivuo et al. 1994). In most surveys, the lead BAF for earthworms exceeds unity only when pH is low (Terhivuo et al. 1994). As for other metals, lead BAFs are typically lower in highly polluted soil. In addition to soil-specific factors, prediction of BAFs for lead in earthworms is complicated by the existence of significant interspecific differences among earthworms exposed to the same soils (Terhivuo et al. 1994).

Ma et al. (1983) and Corp and Morgan (1991) have developed regression equations for predicting lead BAFs in earthworms. However, the equations supporting their data are dependent on pH, organic matter, and calcium concentration. Data on these characteristics are presently lacking for INEEL soils. Until they are available, the following equation from Corp and Morgan (1991), which requires pH and concentration of lead in soil and provides a good fit to the data (r^2 = 93.3) may be used:

$$\log [Lead]_{worm} = 2.65 + 0.897 \times \log [Lead]_{soil} - 3.56 \times \log\ pH \quad \text{(D3-2-11)}$$

$$Earthworm\ BAF_{lead} = \frac{[Lead]\ in\ worm\ tissue}{[Lead]\ in\ soil} \quad \text{(D3-2-12)}$$

As shown in Table D3-2-9, given the pH ranges identified at the INEEL (Martin et al. 1992) and the concentrations of lead in the soil (13 to 72 mg/kg), the earthworm BAFs developed using this equation will range from 0.05 up to 0.23.

Values derived from this equation agree well with field data reported by Beyer et al. (1990) (0.27 to 0.32 at soil lead concentrations of 21 to 336 mg/kg dry weight).

Hopkin et al. (1993) developed regression equations for lead uptake in the terrestrial woodlice (isopods) *Porcellio scaber* and *Oniscus asellus*. The following equation for *O. asellus* yields slightly higher BAFs, and so is recommended as conservative for use at INEEL:

Table D3-2-9. BAFs for lead in terrestrial ecosystems.

Taxonomic Group	BAF
Earthworm	0.18[a]
Arthropod	0.29[b]
Small mammal (*Talpa europea*)[c]	0.6

a. Regression equation from Corp and Morgan (1991) as discussed in text. Soil pH values at various locations on the INEEL ranged from 5.25 to 8.78 (Martin et al. 1992).

b. Regression equation from Hopkin et al. (1993) as discussed in text.

c. Based on the geometric mean kidney:soil lead ratio reported by Ma (1987).

$$log\ [Lead]_{arthropod} = 0.842 \times log\ [Lead]_{soil} - 0.507 \qquad \text{(D3-2-13)}$$

$$Arthropod\ BAF_{lead} = \frac{[Lead]\ in\ arthropod}{[Lead]\ in\ soil} \qquad \text{(D3-2-14)}$$

Given the concentrations of lead in the soil the arthropod BAFs developed using this equation will range up to 0.290 (in mg/kg dry weight). These BAF values agree with field data reported by Janssen and Hogervorst (1993) (0.01 to 0.43).

Tissue concentrations of lead in insectivorous small mammals generally correlate better with ambient lead concentrations and are higher than those of herbivores (e.g., Beardsley et al. 1978; Ma 1987; Ma et al. 1991). A geometric mean lead BAF of 0.08 for the house mouse *M. musculus* can be calculated from the Beyer et al. (1990) data. Whole-body BAFs were not located for insectivorous small mammals, but geometric mean BAFs of 0.6 and 0.2 were calculated for lead in kidney and liver of the mole *Talpa europea* (Ma 1987). Lead concentrations in these tissues were much higher in the shrew (*S. araneus*) than the vole (*M. agrestis*) from the same area (Ma et al. 1991). In the absence of more specifically applicable data, a highly conservative small mammal BAF for lead can be estimated as 0.6 based on the kidney:soil ratio calculated from Ma's (1987) data. A BAF was used for all functional groups to be protective.

D3-2-3.1.8 Mercury

Large differences in both bioconcentration and toxicity of organic and inorganic mercury have been observed in aquatic ecosystems. While methylation of inorganic mercury by methanogenic bacteria is common in aquatic sediments and greatly facilitates metal uptake, the degree of methylation occurring in terrestrial environments is unclear. The mercury present at INEEL was conservatively considered to be entirely organic for purposes of TRV development. To avoid overconservatism, mercury in INEEL soils will be considered to be inorganic for purposes of BAF development.

Romijn et al. (1991) used available data to calculate a geometric mean BAF of 0.4 for inorganic mercury in earthworms. This value also provides a conservative estimate of BAF for other soil-dwelling macroinvertebrates.

Little information regarding bioaccumulation of mercury by other organisms was located. Bull et al. (1977) examined concentrations of mercury in various tissues of woodmice (*Apodemus sylvaticus L.*) and bank voles (*Clethrionomys glareolus Schr.*) collected near a chloralkali plant (mercury contamination ranges of 0.69 to 12.6 mg/kg dry weight) and in an uncontaminated reference area (mercury concentration ranged from 0.04 to 0.19 mg/kg dry weight). As observed with other metals, the BAFs were considerably higher in the control than in the affected area, i.e., uptake decreased with increasing ambient concentration.

Because mercury concentrations in certain areas of INEEL are greater than background, BAFs calculated in the Bull et al. (1977) study (as summarized in Tables D3-2-10) will be used in the ERA analysis. BAFs for the woodmouse tissues ranged from 0.3 in liver to 1.3 in muscle, while those in bank voles ranged from 0.2 in brain to 1.2 in hair. Geometric mean BAFs calculated for all tissues examined were 0.7 and 0.4 for woodmice and bank voles, respectively, will be used for the appropriate INEEL receptors.

D3-2-3.1.9 Zinc

Like chromium and copper, zinc is an essential trace element for many organisms. As a result, it has received relatively little attention as a potential ecological toxicant in terrestrial ecosystems. Estimated BAFs for zinc in macroinvertebrates and small mammals are presented in Table D3-2-11.

As reported for other metals, zinc BAFs in earthworms appear to be inversely dependent on soil concentration. Van Gestel et al. (1993) reported that the earthworm (*Eisenia andrei*) was able to regulate its body concentration of zinc (around 100 mg zinc/kg tissue) at soil concentrations up to 560 mg/kg. Higher "maintenance" levels in tissues were observed in other species (e.g., Ma et al. 1983; Kruse and Barrett 1985; Beyer et al. 1990). Like cadmium, zinc uptake by earthworms is influenced by soil pH (Ma et al. 1983; Corp and Morgan 1991). However, the available regression equations do not adequately

Table D3-2-10. Mean BAFs for mercury in small mammal tissues.[a]

	BAFs	
Tissue	Woodmouse	Bank Vole
Brain	0.7	0.2
Hair	1	1.2
Kidney	0.7	0.5
Liver	0.3	0.2
Muscle	1.3	0.4
Geometric mean	0.7	0.4

a. Data from Bull et al. (1977).

Table D3-2-11. BAFs for zinc in terrestrial ecosystems.

Taxonomic Group	BAF
Earthworm[a]	
~ 1 mg/kg zinc in soil	72
~100 mg/kg zinc in soil	1.3
~500 mg/kg zinc in soil	0.2
Arthropod	0.83[b]
Small mammal[c]	0.7

a. Data from Beyer et al. (1990) and van Gestel et al. (1993).

b. Calculated using the regression equation from Hopkin et al. (1993), as discussed in text.

c. Data from Beyer et al. (1990).

reflect the regulation of zinc concentration evident in field data from several sources. Van Gestel et al. (1993) reported a zinc BAF of 72 at a soil zinc concentration of 1.4 mg/kg. At soil zinc concentrations of approximately 90 to 100, Van Gestel's (1993) and Beyer's groups (1990) reported BAFs of around 1.3. Similarly, BAFs of approximately 0.2 were observed by both groups at soil zinc concentrations of 560 to 570 mg/kg. Zinc BAFs for earthworms should be selected from these ranges on the basis of site-specific soil concentrations (Table D3-2-11).

Several authors have shown a negative dependence of zinc BAF on soil concentrations in arthropods as well (Lindqvist 1992; Janssen and Hogervorst 1993; Hopkin et al. 1993). The regression equations developed by Hopkin et al. (1993) for the terrestrial woodlice (isopods *P. scaber* and *O. asellus*) are representative of these data. The equation for *P. scaber* yields slightly higher BAFs:

$$log\ [Zinc]_{arthropod} = 0.274 \times log\ [Zinc]_{soil} + 1.890 \qquad \text{(D3-2-15)}$$

$$Arthropod\ BAF_{zinc} = \frac{[Zinc]\ in\ arthropod}{[Zinc]\ in\ soil} \qquad \text{(D3-2-16)}$$

As shown in Table D3-2-11, given the concentrations of zinc in the soil, the arthropod BAFs developed using this equation will range up to 0.83 (in mg/kg dry weight).

A study of zinc accumulation in the organs of granivorous and insectivorous small mammals exposed to sewage sludge containing high concentrations of zinc (and other metals), showed some increase with exposure but no pathological effects (Hegstrom and West 1989). Beyer et al. (1990) reported BAFs for the house mouse (*M. musculus*) of 0.4 to 1.2 exposed to soil concentrations of 74 to 240 mg/kg, with a general trend of inverse relationship to soil concentration. The geometric mean of these data, 0.7, is recommended for use at INEEL where soil concentrations are compatible (Table D3-2-11). Data are presently lacking to evaluate BAFs at higher soil concentrations. The homeostatic regulation of zinc in most organisms suggests that BAFs will decrease at higher soil concentrations.

D3-2-4. REFERENCES

Ainsworth, N., 1990a, "Distribution of antimony in contaminated grassland: 1. Vegetation and soils," *Environmental Pollution*, 65(1):65–78.

Ainsworth, N., 1990b, "Distribution of Antimony in Contaminated Grassland: 2. Small Mammals and Invertebrates," *Environmental Pollution*, 65(1):79–87.

Anthony, R. G. and R. Kozlowski, 1982, "Heavy Metals in Tissues of Small Mammals Inhabiting Waste-Water-Irrigated Habitats," *Journal of Environmental Quality*, 11(1):20–2.

Back, H., 1990, "Epidermal Uptake of Lead, Cadmium, and Zinc in Tubificid Worms," *Oecologia*, 85(2):226–232.

Baes, C. F., III, et al, 1984, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*, ORNL-5786, U.S. Department of Energy, Oak Ridge National Laboratory, Oak Ridge, TN.

Beardsley, A., et al, 1978, "Use of the Field Vole (M. Agrestis) for Monitoring Potentially Harmful Elements in the Environment," *Environmental Pollution*, 16(1):65–72.

Beyer, W. N., G. Miller, and J. W. Simmers, 1990, "Trace Elements in Soil and Biota in Confined Disposal Facilities for Dredged Material," *Environmental Pollution*, 65(1):19–32.

Beyer, W. N., R. L. Chaney, and B. M. Mulhern, 1982, "Heavy Metal Concentrations in Earthworms from Soil Amended with Sewage Sludge," *Journal of Environmental Quality*, 11(3):381–385.

Bull, K. R., et al, 1977, "Mercury Concentrations in Soil, Grass, Earthworms and Small Mammals Near an Industrial Emission Source," *Environmental Pollution*, 12(2):1977 135–140.

Corp, N. and A. J. Morgan, 1991, "Accumulation of Heavy Metals from Polluted Soils by the Earthworm, Lumbricus rubellus: Can Laboratory Exposure of Control Worms Reduce Biomonitoring Problems?" *Environmental Pollution*, 74(1):39–52.

Eisler, R., 1987, *Mercury Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, U.S. Fish and Wildlife Service Biological Report, 85(1.10).

Elinder, C. G. and M. Nordberg, 1985, "Metallothionein", in: Friberg, L., Elinder, C. G., Kjellstrøm, T. and Nordberg, G. F. (eds.) *Cadmium and Health: A Toxicological and Epidemiological Appraisal*, CRC Press, Boca Raton, Florida, pp. 65–80.

Environmental Protection Agency (EPA), 1989, "Risk Assessment Methodology Environmental Impact Statement (NESHAPS) for Radionuclides Background Information Document Volume 1," EPA/520/1-89-005, September.

Hegstrom, L. J. and S. D. West, 1989, "Heavy Metal Accumulation in Small Mammals Following Sewage Sludge Application to Forests," *Journal of Environmental Quality*, 18:345–349.

Hopkin, S. P., D. T. Jones, and D. Dietrich, 1993, "The Isopod Porcellio Scaber as a Monitor of the Bioavailability of Metals in Terrestrial Ecosystems: Towards a Global "Woodlouse Watch" Scheme," *Science of the Total Environmental*, Supplement: 257–365.

Hopkin, S. P., 1990, "Species-Specific Differences in the Net Assimilation of Zinc, Cadmium, Lead, Copper and Iron by the Terrestrial Isopods *Oniscus asellus* and *Porcellio scaber*," *Journal of Applied Ecology*, 27:460–474.

Hunter, B. A., M. S. Johnson, and D. J. Thompson, 1987, "Ecotoxicology of Copper and Cadmium in a Contaminated Grassland Ecosystem III. Small Mammals," *Journal of Applied Ecology*, 24:601–614.

International Atomic Energy Agency (IAEA), 1994, "Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments," Technical Reports Series No. 364.

Janssen, M. P. M. and R. F. Hogervorst, 1993, "Metal Accumulation in Soil Arthropods in Relation to Micro-nutrients," *Environmental Pollution*, 79(2):181–189.

Janssen, M. P. M., et al, 1991, "Comparison of Cadmium Kinetics in Four Soil Arthropod Species," *Archives of Environmental Contamination and Toxicology*, 20(3):305–312.

Kalas et al., 1995, "Metals and Selenium in Wild Animals from Norwegian Areas Close to Russian Nickel Smelters", *Environmental Monitoring and Assessment*, **36:**251–270.

Kruse, E. A., and Barrett, G. W., 1985, Effects of Municipal Sludge and Fertilizer on Heavy Metal Accumulation in Earthworms. *Environ Pollut Ser A Ecol Biol* 38: 235–244.

Laskowski, R., 1991, "Are the Top Carnivores Endangered by Heavy Metal Biomagnification?" *Oikos*, 60(3):387–390.

Laurinolli and Bendell-Young, 1996, "Copper, Zinc, and Cadmium Concentrations *in Peromyscus maniculatus* Sampled Near an Abandoned Copper Mine," *Archives of Environmental Contamination and Toxicology*, 30, 481–486.

Lindqvist, L., 1992, "Accumulation of Cadmium, Copper, and Zinc in Five Species of Phytophagous Insects," *Environmental Entomology*, 21(1):160–163.

Ma, W. C., 1982, "The Influence of Soil Properties and Worm-Related Factors on the Concentration of Heavy Metals in Earthworms," *Pedobiologia*, 24:109–119.

Ma, W. C., 1987, "Heavy Metal Accumulation in the Mole, Talpa Europea, and Earthworms as an Indicator of Metal Bioavailability in Terrestrial Environments," *Bulletin of Environmental Contamination and Toxicology*, 39(6):933–938.

Ma, W. C., W. Denneman, and J. Faber, 1991, "Hazardous Exposure of Ground-Living Small Mammals to Cadmium and Lead in Contaminated Terrestrial Ecosystems," *Archives of Environmental Contamination and Toxicology*, 20(2):266–270.

Ma, W., et al, 1983, "Uptake of Cadmium, Zinc, Lead, and Copper by Earthworms Near a Zinc-Smelting Complex: Influence of Soil pH and Organic Matter," *Bulletin of Environmental Contamination and Toxicology*, 30(4):424–427.

Markham, O. D., D. K. Halford, and R. E. Autenrieth, 1980, "Strontium-90 Concentrations in Pronghorn Antelope Bones Near a Nuclear Fuel Reprocessing Plant," *Health Physics* 38:811–816.

Martin et al., 1992, *Preliminary Assessment of Surface Soils At Active EG&G Idaho Facilities Data Document*, EGG-ESQ-9225, Revision 1, May.

Montgomery, J. H. and L. M. Welkom, 1990, *Groundwater Chemicals Desk Reference*, Lewis Publishers, Chelsea, MI.

National Council on Radiation Protection and Measurements (NCRP), 1984, *Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment*, NCRP Report No. 76.

Nellessen, J. E. and J. S. Fletcher, 1993, "Assessment of Published Literature Pertaining to the Uptake/Accumulation, Translocation, Adhesion and Biotransformation of organic chemical by Vascular Plants", *Environmental Toxicology and Chemistry*, Vol. 12, pp. 2045–2052.

Ng, Y.C., C.S. Colsher, and S.E. Thompson, 1979, "Transfer Coefficients for Terrestrial Foodchain - Their Derivation and Limitations," CONF-7810179-1.

Ng, Y.C., C.S. Colsher, and S.E. Thompson, 1979, Transfer Coefficients for Terrestrial Foodchain - Their Derivation and Limitations, Paper presented at 12th Annual Conference of the German-Swiss Fachverban fur Strahlenschultz, Norderney, Germany, 2–6 October, 1978.

Outridge, P. M. and A. M. Scheuhammer, 1993, "Bioaccumulation and Toxicology of Chromium: Implications for Wildlife," *Review of Environmental Contamination and Toxicology*, 130:31–77.

Romijn, C. A. F., et al, 1991, *Presentation of a General Algorithm for Effect-Assessment on Secondary Poisoning. II Terrestrial Food Chains*, Bilthoven, the Netherlands.

Scanlon, P. F., 1987, "Heavy metals in small mammals in roadside environments implications for food chains," *Science of the Total Environment*, 59:317–324.

Stafford, E. A. 1988, *Comparative Heavy Metal Uptake by Soil-Dwelling Invertebrates and the Bioassay Earthworm Eisenia foetida*, Harpenden, Herts, England.

Taylor, F. G. Jr. P. D. Parr, 1978, "Distribution of chromium in vegetation and small mammals adjacent to cooling towers," *Journal of the Tennessee Academy of Science*, 53(3):87–91.

Terhivuo, J., et al, 1994, "Pb uptake by ecologically dissimilar earthworm (Lumbricidae) species near a lead smelter in South Finland," *Environmental Pollution*, 85:87–96.

Travis, C. C. and A. D. Arms, 1988, "Bioconcentration of Organics in Beef, Milk, and Vegetation," Environmental Science and Technology, 22:271–274.

Van Gestel, C. A. M., E. M. D. Van Breemen, and R. Baerselman, 1993, "Accumulation and elimination of cadmium chromium and zinc and effects on growth and reproduction in *Eisenia andrei* (Oligochaeta, Annelida)," *Science of the Total Environment*, Supplement:585–597.

VanHorn, R. L., N. L. Hampton, and R. C. Morris, 1995, *Guidance Manual for Conducting Screening Level Ecological Risk Assessments at the INEL*, INEL-95/0190, Revision 1, June.

Appendix D3
Attachment 3

Plant Uptake Factors

CONTENTS

TABLES

Appendix D3
Attachment 3

Plant Uptake Factors

D3-3-1. INTRODUCTION

The purpose of this section is to document and summarize the selection of literature based plant uptake factors (PUFs) for use in the ecological risk assessment (ERA) exposure modeling. Both organic and inorganic contaminants have been identified on the INEEL. The approach for selecting PUFs for use at the INEEL was different for each type of contaminant as presented in the next two sections.

D3-3-1.1 Plant Uptake Factors for Inorganics

The overall summaries and the values provided by such comprehensive papers such as Baes et al. (1984), IAEA (1994), Ng et al. (1979), and EPA (1989) were useful in selecting PUFs. These studies are well documented and accepted by the decision-makers. Additional studies on native or other grass PUFs identified were given the highest priority, since several investigators have noted the highest Pu concentration in native grasses (Hakonson 1975). This may be true for other contaminants as well since it is noted that the physical structure of the roots of grasses and/or the position within the soil profile are more favorable for uptake (Hakonson 1975).

Attached are two documents produced in support of the evaluation of PUFs for use with native species at the INEEL. One is a letter report documenting a recent review and update of previous work (Attachment D3-3-A). Later an additional evaluation of PUFs for Cs-137, Sr-90, and Tc-99 was performed and is included in Attachment D3-3-B. These were used as the starting point for the effort to locate and identify applicable PUFs for native plant species. This effort provides a good comprehensive reference list for radionuclides.

Table D3-3-1 provides a summary of published values for plant/soil concentration ratios for inorganic contaminants. This summary is based on chemical element rather than specific radionuclide; and as such, ignores any potential isotope effects. Four publications (Baes et al., 1984; IAEA, 1994; EPA, 1989; Ng et al., 1979) provide the focus for these values. Concentration rates (CRs) provided by other publications evaluated are provided in a separate column. The best estimate for use in INEEL ERAs is provided for each element.

D3-3-1.2 Plant Uptake Factors for Organics

The PUFs for organics were calculated using an allometric equation presented in Travis and Arms (1988). This equation is as follows:

$\log PUF = 1.588 - 0.578 \log K_{ow}$.

Log partitioning coefficients (K_{ow}s) were taken from Montgomery and Welkom (1990).

Table D3-3-1. Plant uptake factors used in risk assessment at the INEEL for inorganics.

Element	Baes et al. (1984)	EPA (1989)	IAEA (1994)	Ng et al. (1979)	Other Sources	Suggested for Crops	Suggested, INEEL Native Plants
Aluminum (Al)	4.0E-03	—	—	—	2.2E-03 (Dreesen et al. 1979)	4.0E-03	4.0E-03
Americium (Am)	5.5E-03	5.5E-03	2.2E-05 to 2.2E-03 (carrot)	—	1.7E-02 (Romney et al. 1976) 1.6E-04 (Schulz et al. 1977) 2.1E-03 (Price, 1973)	2.2E-03	1.7E-02 (Romney et al. 1976)
Antimony (Sb)	2.0E-01	2.0E-01	5.6E-04 (root)	—	1.2E-01 (Dreesen et al. 1979)	5.6E-04	2.0E-01
Arsenic (As)	4.0E-02	4.0E-02	—	—	1.3E-02	4.0E-02	4.0E-02
Barium (B)	1.5E-01	1.5E-01	3.0E-02 (not specified)	5.1E-02 (1.8E-02 to 8.3E-02)	—	3.0E-02	1.5E-01
Beryllium (Be)	1.0E-02	1.0E-02	—	—	—	1.0E-02	1.0E-02
Cadmium (Cd)	5.5E-01	5.5E-01	—	—	—	5.5E-01	5.5E-01
Calcium (Ca)	3.5	3.5	—	—	2.7E-01 (Dreesen et al. 1979)	3.5E+00	3.5E+00
Carbon (C)	—	—	—	—	1(EPRI, 1990)	1	1
Cerium (Ce)	1.0E-02	1.0E-02	3.0E-02 (not specified)	4.5E-02 (2.5E-03 to 1.9E-01)	—	3.03-02	3.03-02
Cesium (Cs)	8.0E-02	8.0E-02	5.0E-03 TO 5.3E-01 (grass)	8.9E-02 (3.8E-03 to 5.7E-01)	5.3E-02 (Routson, 1973)	4.6E-01 (mixed greens [IAEA, 1994])	5.3E-01
Chlorine (Cl)	70	—	—	—	—	70	70
Chromium (Cr)	7.5E-03	7.5E-03	1.0E-03 (not specified)	2.9E-02 (3.6E-03 to 8.5E-02)	1.9E-01 (Dreesen et al. 1979)	1.0E-03	1.9E-01

Table D3-3-1. (continued).

Element	Baes et al. (1984)	EPA (1989)	IAEA (1994)	Ng et al. (1979)	Other Sources	Suggested for Crops	Suggested, INEEL Native Plants
Cobalt (Co)	2.0E-02	2.0E-02	3.73E-03 to 1.1 (alfalfa)	2.4E-01 (8.0E-03 to 1.1)	1.0E-01 (Dreesen et al. 1979)	2.9E-01 (spinach, [IAEA, 1994])	1.1E+00
Copper (Cu)	4.0E-1	4.0E-1	8.0E-01	6.7E-01 (5.1E-02 to 1.9)	3.6E-01 (Dreesen et al. 1979)	8.0E-01	8.0E-01
Curium (Cm)	8.5E-04	8.5E-04	2.1E-05 to 1.3E-03 (mixed)	—	4.2E-03 (Price, 1973)	1.3E-03	1.3E-03
Europium (Eu)	1.0E-02	1.0E-02	—	—	—	1.0E-02	1.0E-02
Hydrogen (H)	—	0	—	—	0 (EPRI, 1990)	0	0
Iodine (I)	1.5E-01	1.0	2.0E-02 to 3.4E-03 (grass)	3.1E-01 (1.5E-02 to 1.9)	—	2.0E-02	3.4E-03
Iron (Fe)	4.0E-03	4.0E-03	4.0E-03 (not specified)	1.2E-02 (6.9E-4 to 4.7E-02)		4.0E-03	4.0E-03
Lanthanum (La)	1.0E-02	—	3.0E-05 to 1.0E-03 (carrot)	—	—	1.0E-03	1.0E-02
Lead (Pb)	4.5E-02	4.5E-02	1.1E-03 to 2.0E-02 (root)	—	—	2.0E-02	2.0E-02
Manganese (Mn)	2.5E-01	2.5E-01	4.7E-02 to 9.8 (alfalfa)	2.6 (4.2E-02 to 2.0)	—	1.9E+00 (carrot, [IAEA, 1994])	9.8E+00
Mercury (Hg)	9.0E-01	9.0E-01	—	—	—	9.0E-01	9.0E-01
Molybdenum (Mo)	2.5E-01	2.5E-01	8.0E-01 (not specified)	4.5 (2.3E-02 to 38)	2.5	8.0E-01	8.0E-01
Neodymium (Nd)	1.0E-02	1.0E-02	2.0E-02 (not specified)	—	—	2.0E-02	2.0E-02
Niobium (Nb)	2.0E-02	2.0E-02	1.7E-02 to 5.0E-02 (rape)	—	—	5.0E-02	5.0E-02

Table D3-3-1. (continued).

Element	Baes et al. (1984)	EPA (1989)	IAEA (1994)	Ng et al. (1979)	Other Sources	Suggested for Crops	Suggested, INEEL Native Plants
Nickel (Ni)	4.0E-02	6.0E-02	3.0E-2 to 5.1E-01 (clover)	7.4E-02 (5.7E-03 to 5.5E-01)	3.7 (Dreesen et al. 1979)	6.0E-02	6.0E-02
Neptunium (Np)	1.0E-01	1.0E-01	2.7E-03 to 1.1E-01 (leek)	1.9 (3.7E-01 to 3.5)	2.8E-01 (Price, 1973)	1.0E-01	1.0E-01
Plutonium (Pu)	4.5E-04	4.5E-04	8.6E-06 to 4.4E-03 (carrot)	—	1.1E-03 (Romney, et al. 1976) 2.3E-05 (Schulz, 1976) 3.1E-04 (Price, 1973)	4.4E-03	4.4E-03 (IAEA, 1994; Romney et al. 1976)
Polonium (Po)	2.5E-03	2.5E-02	1.2E-03 to 7.0E-02	—	—	7.0E-02	7.0E-02 (IAEA, 1994)
Praseodynium (Pr)	1.0E-02	1.0E-02	2.0E-02 (not specified)	—	—	2.0E-02	2.0E-02 (IAEA, 1994)
Radium (Ra)	1.5E-02	1.5E-02	1.1E-03 to 1.0E-01 (collard)	—	3.1E-02 (Dreesen, et al. 1979) 2.6E-02 (sagebrush, [Simon et al. 1986])	1.0E-01	2.6E-02 (Simon, et al. 1986)
Rhodium (Rh)	1.5E-01	1.5E-01	9.0E-01 (not specified)	—	—	9.0E-01	9.0E-01 (IAEA, 1994)
Rubidium (Rb)	1.5E-01	1.5E-01	9.0E-01 (not specified)	1.2 (3.3E-01 to 2.0)	—	9.0E-01	9.0E-01 (IAEA, 1994)
Ruthenium (Ru)	7.5E-02	7.5E-02	5.0E-03 to 2.0E-01 (cabbage)	1.4E-01 (1.8E-03 to 5.6E-01)	—	2.0E-01	2.0E-01 (IAEA, 1994)
Selenium (Se)	2.5E-02	2.5E-02	—	—	1.7 (Dreesen, et al. 1979)	2.5E-02	2.5E-02 (Baes et al. 1984)
Silver (Ag)	4.0E-01	4.0E-01	2.7E-04 TO 1.5E-01 (not specified)	—	—	1.5E-01	4.0E-01 (Baes et al. 1984)
Sodium (Na)	7.5E-02	7.5E-02	3.0E-01 (not specified)	7.0E-02 (no range)	—	3.0E-01	3.0E-01 (IAEA, 1994)

Table D3-3-1. (continued).

Element	Baes et al. (1984)	EPA (1989)	IAEA (1994)	Ng et al. (1979)	Other Sources	Suggested for Crops	Suggested, INEEL Native Plants
Strontium (Sr)	2.5	2.5	2.0E-02 (1.7 fodder) to 3.0 (green veg.)	3.5 (1.2E-01 to 23)	19 (Romney et al. 1976)	3.0E+00	1.9E+01 (Romney et al. 1976)
Technitium (Tc)	9.5	9.5	7.3E-01 (grain) to 2.6E+03 (spinach)	—	54 to 20 (Price, 1973)	2.6 E+03	9.5E+00 (Baes et al., 1984)
Tellurium (Te)	2.5E-02	2.5E-02	7.0 (not specified)	—	—	7.0E+00	7.0E+00
Tin (Sn)	3.0E-02	3.0E-02	—	—	—	3.0E-02	3.0E-02
Thallium (Tl)	4.0E-03	4.0E-03	—	—	—	4.0E-03	4.0E-03
Thorium (Th)	8.5E-04	8.5E-04	3.4E-05 to 3.9E-02 (radish)	—	—	3.9E-02	3.9E-02
Tungsten (W)	4.5E-02	4.5E-02	—	—	—	4.5E-02	4.5E-02
Uranium (U)	8.5E-03	8.5E-03	8.3E-03 to 1.4E-02 (mixed roots)	—	5.5E-03 (Dreesen, et al. 1979)	1.4E-02	1.4E-02
Vanadium (V)	5.5E-03	—	—	—	5.5E-04 (Dreesen, et al. 1979)	5.5E-03	5.5E-03
Yttrium (Y)	1.5E-02	1.5E-02	1.0E-02 (not specified)	—	—	1.0E-02	1.0E-02
Zinc (Zn)	1.5	1.5	5.6E-01 to 35 (potato)	9.3E-01 (1.2E-01 to 4.4)	—	3.5E+01	3.5E+01
Zirconium (Zr)	2.0E-03	2.0E-03	1.0E-03 (not specified)	8.1E-02 (4.4E-02 to 1.2E-01)		1.0E-03	1.0E-03

PUFs outside the range of the Travis and Arms (1988) were assigned values at the limits of the evaluation. Table D3-3-2 presents the values for organics identified as present on the INEEL as calculated using the allometric equation.

Table D3-3-2. Plant uptake factors used in risk assessments at the INEEL for organic contaminants.

CAS #	Contaminant	K_{ow}	Calculated PUF	PUFs used in ERA
75-34-3	1,1-Dichloroethane	6.17E+01	3.57E+00	3.57E+00
75-35-4	1,1-Dichloroethylene	6.92E+01	3.35E+00	3.35E+00
71-55-6	1,1,1-trichloroethane	3.16E+02	1.39E+00	1.39E+00
79-34-5	1,1,2,2-Tetrachloroethane	2.45E+02	1.61E+00	1.61E+00
107-06-2	1,2 Dichloroethane	3.02E+01	5.40E+00	5.40E+00
120-82-1	1,2,4-Trichlorobenzene	2.00E+04	1.26E-01	1.26E-01
99-65-0	1,3 Dinitrobenzene	4.17E+01	4.48E+00	4.48E+00
123-91-1	1,4 Dioxane	1.02E+00	3.83E+01	3.83E+01
78-93-3	2-Butanone	1.82E+00	2.74E+01	2.74E+01
95-49-8	2-Chlorotoluene	2.60E+03	4.11E-01	4.11E-01
71-23-8	2-Propanol	NA	1.00E+00	1.00E+00
1746-01-6	2,3,7,8-Tetrachlorodibenzodioxin	5.25E+06	5.06E-03	5.06E-03
94-75-7	2,4-Dichlorophenoxyacetic acid	6.46E+02	9.20E-01	9.20E-01
1300-71-6	2,4-Dimethylphenol	2.63E+02	1.55E+00	1.55E+00
121-14-2	2,4-Dinitrotoluene	1.00E+02	2.70E+00	2.70E+00
606-2002	2,6-Dinitrotoluene	1.00E+02	2.70E+00	2.70E+00
106-47-8	4-Chloroaniline	6.76E+01	3.39E+00	3.39E+00
106-44-5	4-Methylphenol	8.51E+01	2.97E+00	2.97E+00
59-50-7	4-Chloro-3-methylphenol	9.80E+02	7.23E-01	7.23E-01
83-32-9	Acenaphthene	1.00E+04	1.89E-01	1.89E-01
67-64-1	Acetone	5.75E-01	5.33E+01	5.33E+01
75-05-8	Acetonitrile	4.57E-01	6.09E+01	6.09E+01
107-13-1	Acrylonitrile	1.78E+00	2.77E+01	2.77E+01
120-12-7	Anthracene	2.82E+04	1.04E-01	1.04E-01
11097-69-1	Aroclor 1254	1.07E+06	1.27E-02	1.27E-02
11096-82-5	Aroclor 1260	1.35E+07	2.93E-03	2.93E-03
71-43-2	Benzene	1.32E+02	2.30E+00	2.30E+00
8032-32-4	Benzine	NA	1.00E+00	1.00E+00
56-55-3	Benzo(a)anthracene	3.98E+05	2.25E-02	2.25E-02

Table D3-3-2. (continued).

CAS #	Contaminant	K_{ow}	Calculated PUF	PUFs used in ERA
50-32-8	Benzo(a)pyrene	1.15E+06	1.22E-02	1.22E-02
71-36-3	Butyl Alcohol	NA	1.00E+00	1.00E+00
85-68-7	Butylbenzylphthalate	6.31E+04	6.51E-02	6.51E-02
75-15-0	Carbon disulfide	1.00E+02	2.70E+00	2.70E+00
56-23-5	Carbon tetrachloride	4.37E+02	1.15E+00	1.15E+00
7790-86-5	Cerium chloride	NA	1.00E+00	1.00E+00
67-66-3	Chloroform	9.33E+01	2.81E+00	2.81E+00
57-12-5	Cyanide	NA	1.00E+00	1.00E+00
84-66-2	Diethyl phthalate	3.16E+02	1.39E+00	1.39E+00
78-93-3	Di-2-ethylhexylphthalate	9.50E+03	1.94E-01	1.94E-01
84-74-2	Di-n-butylphthalate	3.98E+05	2.25E-02	2.25E-02
117-84-0	Di-n-octylphthalate	1.58E+09	1.87E-04	1.87E-04
64-17-5	Ethanol	4.79E-01	5.93E+01	5.93E+01
100-41-4	Ethylbenzene	1.41E+03	5.86E-01	5.86E-01
206-44-0	Fluoranthene	7.94E+04	5.70E-02	5.70E-02
86-73-7	Fluorene	1.58E+04	1.45E-01	1.45E-01
50-00-0	Formaldehyde	1.00E+00	3.87E+01	3.87E+01
302-01-2	Hydrazine	8.32E-04	2.33E+03	2.33E+03
7439-97-6	Mercury(Organic)	NA	1.00E+00	9.00E-01
67-56-1	Methanol	NA	1.00E+00	1.00E+00
108-10-1	Methyl isobutyl ketone	NA	1.00E+00	1.00E+00
75-09-2	Methylene chloride	2.00E+01	6.85E+00	6.85E+00
91-20-3	Naphthalene	2.76E+03	3.97E-01	3.97E-01
98-95-3	Nitrobenzene	7.08E+01	3.30E+00	3.30E+00
82-68-8	Pentachloronitrobenzene	2.82E+05	2.74E-02	2.74E-02
87-86-5	Pentachlorophenol	NA	1.00E+00	1.00E+00
85-01-8	Phenanthrene	2.88E+04	1.02E-01	1.02E-01
108-95-2	Phenol	2.88E+01	5.55E+00	5.55E+00
129-00-0	Pyrene	7.59E+04	5.85E-02	5.85E-02
7664-93-9	Sulfuric acid	NA	1.00E+00	1.00E+00
26140-60-3	Terphenyl	NA	1.00E+00	1.00E+00
127-18-4	Tetrachloroethylene	3.98E+02	1.22E+00	1.22E+00

Table D3-3-2. (continued).

CAS #	Contaminant	K_{ow}	Calculated PUF	PUFs used in ERA
108-88-3	Toluene	5.37E+02	1.02E+00	1.02E+00
126-73-8	Tributyl phosphate	NA	1.00E+00	1.00E+00
79-01-6	Trichloroethylene	2.40E+02	1.63E+00	1.63E+00
15625-89-5	Trimethylolpropane-triester	NA	1.00E+00	1.00E+00
1330-20-7	Xylene	1.83E+03	5.04E-01	5.04E-01

D3-3-2. REFERENCES

Baes, C. F. III, R.D. Sharp, A. L. Sjoreen, and R. W. Shor, 1984, "A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture," DE-85-000287, September.

Dreesen, D.R. and M.L. Marple, 1979, "Uptake of Trace Elements and Radionuclides from Uranium Mill Tailings by Four-Wing Saltbush (Atriplex canescans) and Alkali Sacaton (Sporobolus airoides)," LA-UR-79-3045.

Environmental Protection Agency (EPA), 1989, "Risk Assessment Methodology Environmental Impact Statement (NESHAPS) for Radionuclides Background Information Document Volume 1," EPA/520/1-89-005, September.

Hakonson, T. E., 1975, "Environmental Pathways of Plutonium Into Terrestrial Plants and Animals," Health Physics 29:583-588.

International Atomic Energy Agency (IAEA), 1994, "Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments," Technical Reports Series No. 64.

Menzel, R. G., 1965, "Soil-Plant Relationships of Radioactive Elements," Health Physics 11:1325-1332.

Montgomery, J. H. and L. M. Welkom, 1990, *Groundwater Chemicals Desk Reference*, Lewis Publishers, Chelsea, MI.

Ng, Y. C., C. S. Colsher, and S. E. Thompson, 1979, "Transfer Coefficients for Terrestrial Foodchain - Their Derivation and Limitations," CONF-7810179-1.

Price, K.R., 1973, "Tumbleweed and Cheatgrass Uptake of Transuranium Elements Applies to Soil as Organic Acid Complexes," BNWL-1755.

Romney, E.M., A. Wallace, P.A.T. Wieland, and J.E. Kinnear, 1976, "Plant Uptake of Pu-239/240 and Am-241 Through Roots from Soils Containing Aged Fallout Materials," UCLA-1056.

Routson, R.C., 1975, "The Effect of Soil Concentrations on the Tumbleweed Uptake of Sr-90 and Cs-137 from a Burbank Sand," BNWL-1905.

Schulz, R.K., G.A. Tomplins, L. Leventhal, and K.L. Babcock, 1976, "Uptake of Pu and AM by Barley from Two Contaminated Nevada Test Site Soils," J. Environ. Qual. 5 (4):406-410.

Simon, S.L., and L. Fraley, Jr., 1986, "Ra-226 Uptake by sagebrush from Uniform and Non-Uniform Soil Amendments," Env. and Exp.Botany 26(1):81-88.

Travis, C. C. and A. D. Arms, 1988, "Bioconcentration of organics in beef, milk, and vegetation," Environmental Science and Technology, 22:271-274.

Appendix D3
Attachment 3A

Plant Uptake Factors at the Idaho National Engineering and Environmental Laboratory

CONTENTS

TABLES

PLANT UPTAKE FACTORS AT THE IDAHO NATIONAL ENGINEERING AND ENVIRONMENTAL LABORATORY

A Literature Review

G.J. White, R. VanHorn, and G. Harris
December 1996

D3-3A-1. INTRODUCTION

Vegetation potentially affects buried waste disposal systems in at least two different ways: (1) Radionuclides or other contaminants may be taken up by plants from the soil, resulting in migration away from the waste containment transfer to the food chain. As the primary producer in terrestrial food chains, plants contaminated with radionuclides can affect animals higher up in the food chain, including humans. Furthermore, radionuclides taken up by plants are available for further migration away from the waste disposal facility. (2). Plant roots can physically breach the containment system of a waste disposal unit.

Uptake through the roots is only one of several pathways where by radionuclides become associated with plants. Deposition to the plant surfaces occurs in areas where measurable levels of fallout occur. Early studies at the Nevada Test Site (NTS) and elsewhere indicated that vegetation growing in areas of significant radionuclide fallout contained activities approximately equal to those found in the soil. However, radionuclide concentrations vary greatly with plant species and with other factors. Many early laboratory studies on plant uptake reported a great deal of variation in radionuclide uptake rates. The results of plant uptake studies at NTS led Romney et al. (1985) to conclude "that the root uptake pathway contributes very little to the amount of Pu and Am contamination generally found in native vegetation growing in the fallout areas of the safety-shot experiments at the Nevada Test Site...".

Depending on the specific radionuclide involved, as well as various factors associated with the plant and environmental conditions, radionuclides deposited on plant surfaces may enter and become assimilated by the plant. Radioactive contamination on the soil surface may also be resuspended and deposited on the plant surfaces. Plant concentrations of radionuclides will vary depending on the relative importance of these different pathways.

For radionuclides found within the soil, however, plant uptake can represent a significant pathway of contamination into the plant and therefore into the food chain. One commonly used measure of plant uptake is the *Concentration Ratio* (CR), defined as the ratio of the radionuclide activity in the plant material to the activity in the soil within the rooting zone, or:

$$CR = \frac{\textit{Radionuclide activity in plant}}{\textit{Radionuclide activity in soil}} = \frac{\textit{Ci/g oven dry vegetation}}{\textit{Ci/g oven dry soil}}$$

CR values less than 1.0 indicate that the plant does not actively assimilate the radionuclides, whereas CRs of greater than 1.0 indicate that either the plant actively absorbs the radionuclides or stores the nuclides after absorption. The CRs are typically based only on the shoot (leaf and stem) portion of the plant, and not on the root. Although it will not be discussed here, the root CRs are typically far higher than the shoot CRs. This could be due to root surface adsorption of the nuclide rather than true radionuclide uptake into the root. It should be noted that as indicated by Baes et al. (1984), CRs (or PUFs) for certain elements are meaningless. Specifically, CRs cannot be produced for C, H, or O, since these are not taken up through the roots in their elemental form. Because of their chemical inertness, noble gases (e.g., Xe, Ne, Ar) cannot be viewed in terms of CRs.

This high inherent variability of CRs provides a substantial complication in the modeling of radionuclide migration within the biosphere. Documented sources of variability in CRs include factors associated with the specific radionuclide and plant species involved, the chemistry and physical properties of the soil, the chemical form of the radionuclide, and other factors. Many of these variables are highly related. For example, the soil type influences the plant species that can grow on it, the chemical form of the radionuclide, and the relative amount of radioactivity available for plant uptake.

The purpose of this document is to provide a review of the available literature on plant uptake of radionuclides and rooting depths to help determine the potential significance of vegetation on the containment of radioactive wastes at the INEEL. This document represents an expansion of an earlier literature review on uptake factors for transuranic radionuclides in desert vegetation at the NTS (Harris 1989). The emphasis of Harris (1989) was on specific radionuclides, notably Am-241, Np-237, Pa-231, Pu-238, 242, Ra-226, Th-229, 232, and U-233, 238. Most of the literature reviewed in Harris (1989) deals with one or more of these radionuclides, though general radionuclide uptake studies are also included. The general conclusion of Harris (1989) was that, whereas a great deal of literature is available on CRs for Pu (and to a lesser extent, Am), there is a general lack of information on other transuranics.

In the present document, a wider variety of radionuclides are considered, including those associated with low-level radioactive wastes. The focus is on plants native to the INEEL plus some commonly-occurring exotic species. However, much of the literature available on plant uptake is for crop species, rather than native, uncultivated species. Harris (1989) noted that there was an almost complete lack of information on the uptake of radionuclides by plant species native to NTS. This poses problems in terms of predicting radionuclide migration and movement through the food chain. For example, Ng et al. (1979) indicated that predictions should consider interspecific differences in uptake as well as variability between different plant parts, and differences in soil properties. Extrapolations from species to species, radionuclide to radionuclide, or site to site are problematic, but may be necessary given the available information.

In general, measured CR values for transuranic radionuclides are less than 0.01 (Harris 1989). In some cases, however, CRs have been reported as high as 0.28 for uptake of Np-237 in tumbleweed or 0.80 for uptake of Am-241 in wheat with the chelating agent DTPA. The maximum uptake rate reported was 2.48 for Pu with the chelator DTPA present.

This report is broken into four sections. The first section is a review of the general literature related to plant uptake of radionuclides. The second section looks closely at relevant root uptake experiments conducted at a variety of locations. Specific information on plant species found on the INEEL (or similar to plant species found at the INEEL) is considered in the third section. In the summary, these results are related to the study at hand and to the modeling process.

D3-3A-2. FACTORS INFLUENCING UPTAKE OF RADIONUCLIDES BY PLANTS

Few generalities can be made regarding the uptake of radionuclides by plants. This is due to the interdependence of uptake on a wide variety of factors associated with the radionuclide species, the plant species, and the soil/chemical environment in which the plant is growing. Furthermore, consideration for many of these factors has been limited in many of the experimental studies conducted to date. Most of the studies have been done with a limited number of radionuclides, hence information for other radionuclides is lacking (Bernhardt and Eadie 1976).

Also, many uptake studies have been conducted under laboratory controlled conditions and on crops rather than natural vegetation. These experiments have largely involved contamination that is uniformly distributed through the soil, whereas in most locations radioactive contamination is not uniformly distributed through the soil (Bernhardt and Eadie 1976). Because of the artificial conditions under which the experiments are carried out, Schulz (1977) suggests that "extrapolations not be made from specialized plant root uptake experiments to field conditions which govern introduction of (radionuclides) into food chains via this pathway."

Substantial variation in radionuclide uptake and concentration in plants has been observed in studies conducted to date, not only for different radionuclides and plant species, but for the uptake of the same radionuclide in the same plant species. Furthermore, individual plant tissues may concentrate certain radionuclides. Important sources of uptake variation external to the plant include soil chemistry (e.g., acidity, presence of chelators, and other soil characteristics), soil physical properties (e.g., size of colloid containing the radionuclide), and the depth of radionuclide burial. The remainder of this section is broken down by source of uptake variability (not necessarily by order of importance).

D3-3A-2.1 Radionuclide Species

The behavior of radionuclides in the environment is dictated primarily by their chemical properties. For this reason, plant uptake rates show substantial variability between radionuclides, ranging from well over 1 to as low as 10^{-6}. Some radionuclides are isotopes of macronutrients (e.g., N, P, S, K, Mg, and Ca) or micronutrients (e.g., Cu, Zn, Fe, Mn, and B). As such, uptake of these radionuclides may be regulated. Others are isotopes of "building block" materials (i.e., C, H, and O) that form the basis for the carbohydrates and other organic compounds that provide the structure of plants.

In some important instances, the chemistry of nonnutrients may be very similar to that of plant nutrients. Cesium, for example, is a Group IA element, as is K. As such, Cs is a chemical analog of K and tends to act in a similar fashion chemically. Similarly, both Sr and Ra are Group IIA elements, and are therefore chemical analogs of Ca and Mg. Factors responsible for preferential uptake of these plant nutrients may also allow for the preferential uptake of the chemical analogs. Similarly, physiological or biochemical processes that act to concentrate chemical elements in various plant (or animal) tissues may also tend to accumulate analogous radionuclides. In contrast, the transuranic radionuclides discussed by Harris (1989) do not generally have nutrient analogs and maintain somewhat unique chemical properties.

It is generally believed that isotopes of a given element are chemically identical. For example, Nishita (1981) and Schulz and Ruggieri (1981) concluded that there were no differences in plant uptake rates between Pu-238 and Pu-238, 240. However, there is some disagreement as to whether different isotopes of the same element exhibit different uptake rates. For example, Adriano et al. (1981) observed

nearly a 100-fold higher uptake rate for Pu-239, 240 than for Pu-238, implying a substantial "isotopic effect."

To illustrate the range associated with CRs for different radionuclides, Table D3-3A-1 provides a gross summary of reported CRs for some of the more commonly encountered fission products and transuranic radionuclides. As can be seen in Table D3-3A-1, the range of CRs reported is over several orders of magnitude. In general, the transuranic radionuclides generally have low CR values, indicating little biological and/or environmental mobility. Other radionuclides, especially certain important fission products such as Sr-90, exhibit CRs significantly higher than 1, indicating accumulation above soil concentrations. It should be pointed out that the values reported in Table D3-3A-1 and elsewhere in this document refer to concentrations in the aboveground portions of the plants. Several transuranics and other radionuclides may accumulate in much higher concentrations in (or adsorbed to the surfaces of) root tissues.

D3-3A-2.2 Plant Characteristics

Uptake of radionuclides and other contaminants exhibits substantial variability between species. This is due to a variety of morphological and physiological differences between plant species. For example, species with pilose ("hairy") leaves, such as Winterfat (*Ceratoides lanata, a.k.a. Eurotia lanata*) are more effective in trapping atmospheric particles, leading to higher concentrations within the plant. Price (1971) reports that the *Chenopodiaceae* are known to accumulate radionuclides. This is relevant since the Chenopodiaceae make up a large part of the flora of NTS, including saltbush *(Atriplex* spp.), winter fat *(Ceratoides lanata),* and hop sage *(Grayia* spp.).

Other plant species have sticky leaves. The structure of the plant canopy can also influence the interception of particles from atmosphere. Most of the uptake of nutrients and other materials by mosses and lichens is through the leaf surfaces rather than from roots. High absorption of deposited materials including surface-deposited radionuclides has been reported for these plant types (e.g., Lopatkina et al., 1970). Similarly, hydrophytes (plants living within water) have been reported to take up radionuclides more readily than mesophytes or xerophytes (e.g., Nagpal et al., 1974).

Dahlman et al. (1976) found that trees generally absorbed Pu less readily than understory species (both herbaceous and shrub). This may be related either to the higher stature of the trees, allowing greater interception of atmospheric particles, or to a greater rooting depth. The understory species behaved alike and also like field crops grown in the same area. Grasses generally appear to absorb less radionuclides than other vascular plants, while some species, such as tumbleweed, are known for their ability to uptake fission products (Price 1972).

Several studies report relatively high concentrations of radionuclides in the roots, with degree of transport to the shoots varying from radionuclide to radionuclide. For example, Prister and Prister (1970) show a much greater concentration of U in the roots of corn than in the shoots. They also note a slight decrease in shoot uptake as the concentration of U in the roots and soil increases. D'Souza and Mistry (1970) report the ratio of shoot content to total plant content for Th-230 and Ra-226 are 0.12% and 20.62% respectively.

With respect to buried waste disposal sites, the key morphological factor determining radionuclide uptake is rooting depth, which influences the potential rates at which radionuclides are taken up by plants simply by dictating the quantities of contaminants available to the plant. Although information on rooting depths in desert plants is somewhat limited, desert plants exhibit two distinct rooting strategies. The first involves the development of a large tap root capable of infiltrating deeply into the soil to reach the water

Table D3-3A-1. Summary of concentration ratios reported for important radionuclides.

CR Range	Radionuclides
10 to 1,000	Na-22, * questionable
1 to 100	Tc-99, Sr-90
0.1 to 10	Ra-226, I-229, Co-60, Ni-63
0.01 to 1	Cs-134, Cs-137, Be-10, Np-237
0.001 to 0.1	U-234, U-235, U-238
< 0.01	Am-241, Cm-244. Th-228, Pu-238, Pu-239, 240, Pu-241, Sb-125
Not Applicable	C-14, Eu-152, Eu-154, Eu-155, H-3

Similarly from Menzel, 1965, produced the following CR summary

10 to 1000 Strongly	1 to 100 Slightly	0.1 to 10 Not	0.01 to 1 Strongly	< 0.01 Slightly
K	Mg	Ba	Cs	Sc
Rb	Ca	Ra	Be	Y
N	Sr	Si	Fe	Zr
P	B	F	Ru	Ta
S	Se	I		W
Cl	Te	Co		Ce
Br	Mn	Ni		Pm
Na	Zn	Cu		Pb
Li	Mo			Pu
				Sb

table below the plant. The second rooting strategy is to spread diffuse roots near the surface in order to more effectively capture rainwater when it is present as it infiltrates the soil.

Wallace et al. (1980) studied the depths of roots of nine native perennial species (48 individual plants) at the NTS. The species examined included *Atriplex canascens* (fourwing saltbush), *A. confertifolia* (shadscale), *Acamptopappus shockleyi, Larrea tridentata* (creosote bush), *Ephedra nevadensis* (Mormon tea), *Lycium andersonii* (wolfberry), *L. pallidum* (wolfberry), *Krameria parvifolia*, and *Ambrosia dumosa* (burro bush). In virtually all cases, the root systems were distributed entirely within the first 50 cm of soil. However, in this case rooting depth was apparently limited by caliche layers or by unfavorable soil chemistry or soil physics. Hence, the only places where deep roots would be expected are in areas where rain water accumulates. Although there were differences between root morphology among the species, it is not certain that these species represent the root morphologies of all species native to that area. There was no mention of the age or size of the plants studied in the report, nor of how the plants were selected other than that they were positioned away from other plants to avoid mixing roots of different individuals.

Root biomass distribution with depth for these desert shrubs was also reported by Wallace et al. (1980). Considering all species examined, they found that 39% of the total belowground biomass was located within the first 10 cm of soil, while 70% was within the first 20 cm, 86% in the first 30 cm, and 95% in the first 40 cm. Only the two saltbush species studied (*Atriplex canescens* and *A. confertifolia*)

had roots growing below 50 cm, accounting for about 2% of total root biomass of these two species. This may be attributable to the high degree of salt tolerance exhibited by this species.

In an earlier study at the NTS, Wallace and Romney (1972) examined root systems of a number of commonly found shrubs at the NTS. These included *Franseria dumosa* (burro bush), *Hymenoclea salsola, Ephedra nevadensis* (Mormon tea), *Larrea divaricata* (creosote bush), *Eurotia lanata* (winterfat), *Lyclum andersonii* (wolfberry), and *Krameria parvifolia*. Although this study did not specifically examine rooting depth, pictures in the report indicate depths penetrating from 50 to 100 cm for all of these shrubs. However, they note that the caliche hardpan layer in the area studied was as deep as 70 cm and hence concluded that the depth of the root systems would be greater than that found elsewhere at the NTS where the hardpan layer is shallower.

Rooting depths of plants common to Los Alamos National Laboratory (LANL) have also been examined (e.g., Foxx et al., 1984a, 1984b, Foxx and Tierney, 1986, and Tierney and Foxx, 1987). An extensive literature search associated with these publications indicated that the roots of annual grasses are generally restricted to the top 1 m of soil while roots of annual forbs average less than 1 m depth. Perennial grasses and forbs both average slightly over 1-m root depth. Shrubs averaged about 2 m depth and trees only 1.6 m, with maximum reported depths for shrubs of 17 m and for trees of 61 m. The deepest roots observed were for alfalfa (*Medicago sativa*) and one-seed juniper (*Juniperus monosperma*), which were found at depths of 39 and 60 m, respectively.

Some species common to NTS were included in the data reported by the Los Alamos group. Rabbitbrush (*Chrysothamnus nauseosus*) had a maximum observed depth of 2.1 m at LANL, while the literature reports rooting depths up to 4.5 m for this species. For yucca (*Yucca* spp.), roots went as deep as 2.1 m at Los Alamos. Big sagebrush (*Artemisia tridentata*) is reported in the literature to have roots up to nearly 10 m, and globe mallow (*Spheralcea* spp.) has roots up to 4 m in depth. Four-wing saltbush (*Atriplex canescens*) roots have been reported to 7.5 m in depth. Foxx and Tierney (1986), Tierney and Foxx (1987), and Foxx et al. (1984a, 1984b) do not report any root depth information on the dominant plant at the RWMS, creosote bush (*Larrea tridentata*).

Selders (1950) studied absorption of radionuclides by Russian thistle (*Salsola* spp.) at the Hanford Waste Site and believed that the roots of this common species reach over 10 m in depth. Wallace and Romney (1972) also give descriptions of common NTS plants that sometimes include vague mentions of root depth. For creosote bush (*Larrea divaricata*), they claim depth of root corresponds closely to depth of penetrating moisture. In sagebrush (*Artemisia tridentata*), roots tend to grow densely and spread laterally in shallow soils or grow deep into well structured soils. *Krameria parvifolia* is reported to have a very shallow root system. Blackbrush (*Coleogyne ramosissima*) roots penetrate to the hard pan layer and spread in all directions.

Based on Case et al. (1984), the maximum depth of infiltration is about 50 cm at RWMS and the maximum depth of infiltration over the past 10,000 years appears to be about 200 cm. Therefore, for those species studied by Wallace et al. (1980) at the NTS, the roots will not penetrate to the depths that the waste is buried (70 to 120 ft). In any case, since the species studied are the dominant species and since the plants are fairly sparsely distributed, it would seem that significant root penetration by less common species to the waste is unlikely.

However, from a personal communication, Dr. Richard Hunter of Reynolds Electrical and Engineering Company (REECO) reports seeing a root at approximately 7 m in depth at the NTS, which he believed was from a creosote bush. He also reports seeing tiny roots at up to 15 m in depth in trenches at the NTS. Hence, there is some possibility of root penetration of the waste. This suggests the need for further studies on root depth since Wallace et al. (1980) considered only a limited set of plant species as

well as a limited area of study. Furthermore, the plants selected for root depth measurements were "selected to give minimum interference to adjoining shrubs." This implies that inter-plant competitive stress was not a factor for the depths obtained by the individual plants, but this stress may force plants to grow deeper roots.

One limitation of many of the plant uptake studies conducted to date is that they have focused on a limited number of plant species (often only one). In an ecological setting where several plant species are present, the canopy structure is more complex and may therefore be more efficient at trapping atmospheric contaminants. Similarly, a plant community consisting of several different species with different rooting strategies will make more effective use of the available soil volume, potentially resulting in a greater total uptake than observed in a limited laboratory or field study.

D3-3A-2.3 Soil Chemistry

Aside from inherent differences between radionuclides and between plant species, plant uptake of radionuclides is substantially dependent on a variety of related factors associated with soil chemistry. It is generally believed that any feature that increases nuclide solubility in soil or increases plant vigor can be expected to increase plant uptake of radionuclides (Price 1972). The behavior of radionuclides (or other contaminants) in the plant/soil system is therefore strongly dependent on several interrelated aspects of soil chemistry. These include soil pH, oxidation state, presence of natural or artificial chelating agents, nutrient status (including additions of fertilizers), as well as other factors.

D3-3A-2.4 Soil pH

The acidity of the soil has a strong influence on the mobility of radionuclides and other contaminants in the soil, which in turn influences availability of uptake. The degree to which pH affects plant uptake is also related to the chemical properties of the radionuclide. Heavy metals and transuranics tend to adhere to organic matter, but will become increasingly mobile when the soil solution reaches a certain pH. Similarly, Cs and some other cations can become involved in the soil cation exchange complex, with the relative amounts of these materials in the soil solution largely dependent on pH.

Rediske et al. (1955) found that the CR for Pu increased from 10^{-4} to 10^{-3} with a reduction in pH from a neutral 7 to an acidic 4. Wilson and Cline (1966) found Pu uptake from an acid soil was three times that from a calcareous (i.e., basic) soil. Romney et al. (1976) found that soil acidification following the addition of S resulted in a significant increase in the uptake of both Am-241 and Pu-239 ,240, but indicated that acidic edaphic conditions "are unlikely to occur in the soils of aged fallout areas at NTS and TTR (Tonopah Test Range) because of their high buffering capacity."

The addition of lime and associated reduction in pH has been shown to decrease plant uptake rates of Ra-226 and Am-241 (e.g., Adriano et al., 1977; Mistry and Bhujbal, 1973; Hoyt and Adriano, 1976; and Vavilova and Rusanova, 1972). In general, the more acidic the soil the greater the uptake of transuranic and some other radionuclides, although there is a limit to this relationship. If the pH goes too low, uptake will decrease due to damage to the fine roots. However, soil pH does interact with other variables such as DTPA, as reported by Wallace (1972). Au and Beckert (1977) found that lower media pH values increased the absorption of Pu by microorganisms, which may fix the Pu for plant uptake.

Nishita et al. (1981) studied the relationship between soil pH and extractability of Np-237, Pu-239, and Am-241 from various soils. Presumably, the more extractable the radionuclide is, the more available it is for uptake. In general, extraction was high for pH values less than 2. The extractability rapidly

decreased up to between pH 5 and 7. Often they observed another peak for extractability around pH 10. They believe that the high extractability at low pH values is due to the radionuclides being in a free ionic form. As the pH increases, hydrolysis and colloid formation of the radionuclides occurs along with an increase in the sorption to soil particles. The extractability is lowest when the colloids formed are at their lowest solubility. Nishita et al. (1981) suggest that the peak beyond pH 7 may be due to the solubility of organic matter (OM) with which the radionuclides are associated at various pHs.

D3-3A-2.5 Oxidation State

Dahlman et al. (1976) report that the order of uptake for radionuclides (Np > Am > Cm > U > Pu) appears to be related to the order of oxidation state species, V, VI, III, IV. This is in general agreement with Bondietti and Sweeton (1977) who report that plant uptake appears to follow V > III = VI > IV valences. Specifically, U-238, with valence state V is more readily taken up than Th-232 and Pu-239, which have valence states of III and VI, respectively. Hence, Bondietti and Sweeton (1977) also conclude that the relative availability of transuranic elements to plants appears to be relative to the oxidation state present in soil. Jacobson and Overstreet (1948) found the order of uptake for Pu was PuO_2^{+2}(VI) > Pu(IV) > Pu(III).

Differences in radionuclide availability because of different oxidation states may be related to either relative reactivity of radionuclides with soil components or relative radionuclide insolubilities (Bondietti and Sweeton 1977). Dahlman et al. (1976) present data that appear to confirm a relation between oxidation state and sorption of the radionuclide to soil colloids. They show that the order of sorption to clay is Pu(IV) ≈ Th(IV) > U(VI) > Np(V). Those nuclides not sorbed to soil are assumed to be more mobile and available for plant uptake.

Dahlman et al. (1976) caution that Pu and Np may be present in multiple oxidation states. Hence, the above ordering of the radionuclide uptake is only for the oxidation states given. Data for oxidation state II, which corresponds to Ra, is not available. However, from D'Souza and Mistry (1970), Ra is taken up at about 100 times greater rate than Th, which is oxidation state IV. This is roughly the same magnitude of difference between oxidation state V and IV. Pa, for which there is no available information, can have oxidation states of either V or IV; it is not known which oxidation state is stable in soil. Hence, the literature suggests the following ordering of plant uptake for the seven radionuclides of interest: Ra(II) ≈ Np(V) ≈ Pa(V) > Am(III) ≈ U(VI) > Th(IV) ≈ Pu(IV) > Pa(IV).

D3-3A-2.6 Presence of Chelating Agents

A chelating agent is a material that promotes the formation of chelates, which are chemical compounds in which a metallic ion is firmly combined with a molecule by means of multiple chemical bonds. A number of studies have indicated that the presence of chelating agents increases plant uptake of Pu and Am. As with any factor that alters the chemistry of the soil, the effects of chelating agents on plant uptake varies with radionuclide.

DTPA has been the chelator most commonly tested, beginning with the study by Hale and Wallace (1970), which found an increase in Am-241 uptake of two orders of magnitude in bush beans. However, the effect decreased dramatically after 30 days suggesting that the DTPA's effect is short term, either being transformed over time or attached to soil colloids. Lipton and Golden (1976) found an increase of 10 times the uptake of Pu-239 due to addition of DTPA. Romney, et al. (1970) found an increase in the uptake of Pu-239 due to addition of DTPA, and only a slight increase in Pu uptake due to addition of the

chelator EDDHA. McLeod, et al. (1981) found addition of EDTA had little or no affect on the uptake of Pu.

Wallace (1971) found that the effect of DTPA on uptake of Am-241 depended on soil pH such that the greatest concentration of Am-241 in the plants were found in plants grown in soils at pH values giving maximum Am - DTPA stability. Wallace (1972) found that DTPA increased the uptake of Am-241, but that the addition of the chelator RA 157 had little or no effect. Wallace (1972) also notes that root temperature did not affect uptake of Am and hence speculates that the DTPA chelator effect is not related to plant metabolism. However, Wallace et al. (1981) concluded that the increase in shoot CR of Am-241 in bush beans was due to DTPA increasing the transport of Am-241 from the root to the shoot, rather than by increasing total uptake.

Romney et al. (1976, 1978, 1985) found DTPA increased uptake of Am and Pu. They found the effect to be greater in Pu and noted that the chelator effect diminished over successive harvests. However, Romney et al. (1981) reported no effect of DTPA on the uptake of Np-237. Wallace et al. (1977) report that DTPA enhanced the transfer of Am from the roots to the shoots. They also report that an increase in DTPA concentration increases the rate of uptake at rates slightly greater than proportional to change in concentration.

Francis (1973) believes that the most probable mode of Pu entry into food chains leading to man would be that chelated with naturally occurring organic soil components. The chelator likely increases the solubility of the radionuclides for the plants to uptake. Hence, it seems feasible that chelator effects can occur for all transuranics, though it has only been documented for Am and Pu.

D3-3A-2.7 Presence of Fertilizers

Nutrient status has also been shown to influence the uptake of radionuclides from the soil. For example, Sultanbaev (1974) and Ananyan and Avetisyan (1971) found that addition of fertilizers increased uptake of U and Ra, respectively. Other studies have indicated that the uptake of some radionuclides is inversely related to the relative availability of nutrient analogs in the soil. For example, Cs-137 uptake may be enhanced in K-deficient soils, while the uptake of Ra-226 and Sr-90 may increase in Ca-deficient soils.

D3-3A-2.8 Other Soil Chemistry

Other sources of variability in plant uptake of radionuclides due to soil properties such as cation exchange capacity (CEC), percent exchangeable cations, and total organic carbon (TOC) content of the soil have also been documented. Miner and Glover (1974) found significant relationships between the sorption of Pu and groups of chemical and physical characteristics of the soils that are associated either with the ion exchange capabilities of the soils or their acidities. Differences in Np-237, Pu-239, Am-241, and Cm-244 uptake by plants based on the organic acid complex in which the radionuclides were contained in were reported by Price (1973b). Price found that Np and Pu uptake from organic acid complexes such as oxalate or citrate generally was greater than uptake from nitrate, whereas, uptake from Am and Cm organic acid complexes was less than from nitrate forms.

D3-3A-2.9 Physical Properties of the Soil

In addition to the chemical properties of the soil, plant uptake of radionuclides and other contaminants is also related to the physical properties of the soil. Since uptake of many contaminants is

related to the mobility of the contaminant in the soil solution, the moisture content of the soil is also a factor. In desert systems during dry periods, there will be little movement of contaminants both directly due to the lack of moisture and indirectly due to the aquiescence of the vegetation growing in the soil. Soil temperature is also related to plant activity, and therefore to uptake.

Another factor is how the contaminant is distributed within the soil. For example, if Cs-137 is found in large particles in the soil, the activity is not available to the plant. Therefore, CF would appear to be low simply because the form of the contaminant does not allow for it to be taken up. Lipton and Golden (1976) found some effect on Pu-239 uptake of soil colloid size, with smaller colloids resulting in greater uptake rates. Little et al. (1973) reported an increase in plant Pu concentration as the particle size decreased. Nishita and Haug (1981) found that Pu extractability from clay minerals depended on the clay mineral type, clay particle size distribution, and the NO_3^- concentration.

D3-3A-2.10 Site Activities

One problem with definition of CR is that it assumes that the contaminant is distributed uniformly throughout the volume of soil. In most contamination scenarios, this is not an accurate assumption, regardless of whether the contaminant is deposited to the surface or is migrating upward in the soil column from a waste disposal unit. Where contamination is deposited to the surface, this can result in an overestimation of CR because the contamination is restricted to a small portion of the rooting zone (e.g., Baes et al.1984).

In a similar manner, the depth of burial and the degree of upward migration of contaminants will also influence uptake. Lipton and Golden (1976) found some effect of the depth of burial on Pu-239 uptake (shallower depths had greater uptake).

Several of the studies available on radionuclide uptake in crop plants may yield higher CR values than in natural systems because of disturbance to the soil from cultural practices. Pinder et al. (1976) found Pu concentrations to be lower in crop plants than in natural vegetation by a factor of 10^{-1} to 10^{-2}, attributing the difference to soil disturbance from plowing.

D3-3A-2.11 Radionuclide Concentration in Soil

There has been some study of how the concentration of the radionuclide in the soil affects plant uptake. For instance, Prister and Prister (1970) found that the relationship between the accumulation of U-238 by the plants and its content in the medium is a decreasing exponential function. Gilbert and Eberhardt (1976) found that CR values for U-234, U-235, U-236, and U-238 decreased as soil concentration increased. Price (1971), however, reports that plant uptake from a thin band of contaminated soil is greater than the same amount of radionuclide spread throughout the soil (for a fixed volume of soil). Wallace et al. (1977) report results that imply that the CR of Am remained constant regardless of the Am concentration in the soil. Wallace et al. (1981) report a decrease in the CR for Am-241 in the leaves of bush beans and an increase in the CR for stems with a 16-fold increase in soil concentration with no DTPA present. When DTPA was present, the CR was nearly constant. McLeod et al. (1981) found uptake rates for Pu to be independent of soil concentration. From the data reported in Dreesen and Marple (1979) it can be inferred that the CR of both Ra-226 and U was smaller at higher concentrations of the radionuclides in the soil.

D3-3A-2.12 Time

Both radionuclide uptake rates and ultimate concentrations in plants appear to increase with time. Logically, the concentrations of many radionuclides in plant tissues increase with the lifetime of the plant or the plant part, unless the radionuclide is systematically eliminated from the plant. Wheeler and Hanchey (1971) presented evidence that radionuclides are stored in the vacuoles of plant cells, suggesting that perennials may have higher concentrations of radionuclides than annuals. Newbould (1963) found that Pu uptake increased by four times during a 2-year study of the perennial ryegrass. Newbould and Mercer (1961) also reported an increase of Pu uptake by ryegrass for successive harvests (4) during the first year. Romney et al. (1970) found a consistent increase in Pu accumulation in plant tissue during a 5-year cropping sequence. They witnessed a seven-fold increase in uptake over the 5-year period; however, they could only speculate whether this increase could be attributed to development of the root system or Pu becoming more available for root uptake.

Wallace et al. (1977) report that Pu became more available with time for clover grown continuously in potted soil for 5 years. Price (1973a) suggests that the three possible mechanisms that may cause the observed increase in plant uptake with time are formation of organic complexes, concentration buildup at root surfaces, and slow, but continual, uptake by perennials. Au and Beckert (1975) suggest that microbial actions could contribute to the increased rate of plutonium uptake by plants over time.

However, even with annuals, radionuclide concentrations generally increase with time. Morishima, et al. (1976) reported that the U concentration in leaves of radish, pimento, and cucumber plants increased slightly during plant growth.

D3-3A-3. RESULTS OF SPECIFIC STUDIES

D3-3A-3.1 Nevada Test Site (NTS)

A series of studies on plant uptake of transuranic radionuclides at the NTS was conducted by the Nevada Applied Ecology Group (NAEG) during the 1970s and 1980s (e.g., Romney et al. 1976, 1978, 1985). The NAEG studies may represent the largest effort in examining radionuclide contamination in terrestrial environments conducted before the Chernobyl accident. These studies centered on contamination of areas within the NTS resulting from test firings of nuclear devices. The source of radionuclide contamination was from fallout and resuspension of deposited materials rather than hazardous waste burial. Most of the NAEG studies focus solely on Pu, although Am was also considered.

Romney et al. (1976, 1978) reported results from three experiments in which they examined the uptake of Pu and Am from contaminated soils collected from the NTS and Tonopah Test Range. These experiments were conducted in crop plants (barley, alfalfa, and soybean) under greenhouse conditions. In the first of these experiments, the effects of various soil amendments (e.g., N, S, and OM) and the addition of a chelating agent (DTPA) on the uptake of Pu and Am in barley was examined. They also observed the distribution of Pu and Am within the aboveground parts of the barley plants. Their results indicated that CRs for Am were generally an order of magnitude higher than those of Pu. With respect to the chelating agent, they concluded that the addition of DTPA generally increased uptake of both radionuclides. As for the other chemical additions, N fertilizer and OM were not observed to influence uptake rates. However, the addition of S, particularly in combination with DTPA, increased uptake of both Pu and Am. This effect was attributed to the acidification of the soil due to the S addition. Romney et al. (1976, 1978) also concluded that Pu and Am were distributed uniformly throughout the plants, with no plant tissues exhibiting significantly elevated concentrations relative to other tissues, although the barley seed heads generally showed lower concentrations than stems and leaves.

The second experiment conducted at the NTS by the NAEG was similar to the first except that alfalfa was used and within-plant distribution of the radionuclides was not studied. As with barley, DTPA and S applied both separately and together increased uptake of Am and Pu, while N and OM had no effect. They attributed the observed increase of Pu uptake in soil with both OM and DTPA to acidification of the soil caused by a high OM treatment. They further noted that the effect of DTPA appeared to decrease with time.

In the third experiment, the same factors were studied for soybean, along with a comparison of uptake rates from different soils. Results were much the same as for the first two experiments. Differences in uptake rate due to soil source were not quantifiable due to variation of concentration between the soils.

Romney, et al. (1976, 1978) noted that although Am, which is a decay product of Pu, had higher CR values than did Pu for all three plant species, the concentration of Am in the soil (and hence in the plant) was much lower than Pu. The mean Am CR/PU CR ratio was 21.6 for soybean, 9.9 for alfalfa, and 4.2 for barley.

Romney, et al. (1985) reported basically the same results as in their previous reports (Romney, et al., 1976, 1978), but included carrot, bushbeans, and wheat in the study. Not surprisingly, they found that DTPA increased uptake of both Pu and Am. They did not study addition of other soil amendments in this report. They note that Am and Pu concentration in the stems and leaves is about 10 times greater than in grain and fruiting bodies. Observed differences in uptake due to soil source could not be quantified due to high variation. They also report that alternate freezing and thawing of the soils at 7-day intervals did

not increase the leachability of Pu or Am from the soil. Results of these studies are summarized in Table D3-3A-2.

Au (1974) and Au and Beckert (1975) examined the uptake of Pu by soil microorganisms at NTS, and reported that they did not discriminate against uptake of Pu, though he used a very acidic substrate on which to grow the microorganisms. Au (1974) concludes this:

> "suggests that the discrimination against Pu uptake reported for plants is a function of the physical and chemical properties of the Pu contamination and is not necessarily dependent on the discrimination by plants. It is possible that in acid and moist soils, Pu dioxide may be solubilized. If this is true, Pu could probably be readily available for absorption. Differences in Pu absorption by various plant species may, therefore, depend on the nature and extent of toot exudates and soil microorganism interrelationships."

Au and Beckert (1975) hypothesize that:

> "microbial actions could be responsible for, or contribute to, the increased rate of plutonium uptake by plants with increasing time... This would mean that availability to mankind of plutonium deposited in the environment could increase over decades or centuries at an as-yet-unknown rate."

D3-3A-3.2 Hanford

Several other plant uptake studies were conducted during the 1970s at Hanford, although these studies were not as extensive as that conducted at the NTS. As compared with the NTS studies, the studies conducted at Hanford are somewhat more relevant to the INEEL for a couple of reasons. First, the Hanford studies were conducted with respect to radioactive contamination arising from leakage of buried wastes. Second, the plants found in the Hanford area are somewhat more typical of what is found at the INEEL. As with the NTS studies, however, the work conducted at Hanford also focused on transuranic radionuclides.

Price (1972, 1973b) conducted a series of studies on plant uptake of radionuclides at Hanford on tumbleweed (*Salsola* spp.) and cheatgrass (*Bromus tectorum*). Both of these species are nonnative invaders that are now distributed widely throughout the western U.S. due to their ability to grow on disturbed soil. Tumbleweed is an annual, noxious weed that is known to be capable of taking up large quantities of fission products (Price 1972; Selders 1950). Cheatgrass, like other annual grasses, is not known for its ability to accumulate radionuclides. The shortcomings associated with using these species in modeling efforts is that neither are perennial. As such, they lack the deep root systems necessary to reach the confinement barriers of waste disposal sites, and they are incapable of accumulating radionuclides over large time spans. Furthermore, these species do not likely represent the characteristics of native vegetation with respect to radionuclide uptake. However, they may provide a reasonable representation of the extremes of plant uptake of radionuclides in this region.

Price (1972) studied the uptake of Np-237, Pu-239, Am-241, and Cm-244 in a nitric acid complex. The nitric acid solution acidified the soil to a pH of 5.1, as compared with a pretreatment pH of 7.8. Plants were grown in a soil classified as Burbank loamy sand collected at the Hanford site. These soils were assumed to be "clean" in that they were not contaminated by radionuclides. The radionuclides were added to the soil as spiked solutions. Price (1972) cites previous studies in which it was shown that Am and Pu were held tenaciously by the soil if salts, acids, detergents, organic compounds, etc., were absent

Table D3-3A-2. Maximum Cr values from the experiments of Romney and Wallace.

	Control		+DTPA	
	Pu	Am	Pu	Am
Wheat[a]	1.76 E-3	3.85 E-2	1.35 E-2	8.00 E-1
Alfalfa[a]	8.04 E-4	1.17 E-1	2.61 E-2	8.07 E-2
Bushbean[a]	7.24 E-3	1.77 E-1	1.16 E-2	4.92 E-1
Carrot[a]	6.20 E-2	7.65 E-2	6.79 E-2	1.23 E-1
Soybean[a]	1.10 E-3	6.60 E-3	1.70 E-2	3.70 E-1
Barley	4.1 E-4(C)[b]	5.8E-4(S)	5.7E-3(S)	4.2E-3(S)
Alfalfa	1.3 E-4(O)	1.lE-3(S)	3.6E-4(S)	1.9E-2(S)

a. Plants were either grown in soil with DTPA or without DTPA. No other soil amendments were added.

b. (C) = Control soil (no soil amendment), (S) = Sulfur added to soil, (O) = Organic matter added to soil.

and only water used. All four radionuclides are assumed to form insoluble hydroxy-or oxy-radiocolloids at typical soil pHs. Thus it was concluded that the movement of the radionuclides in the soil due to daily irrigation would be negligible. The plants were then grown from seed in the soil under climatically controlled conditions.

The results of Price's first study showed relatively large CRs for Np-237 in both tumbleweed and cheatgrass, with considerably lower CRs for Pu-239, Am-241, and Cm-244 (Table D3-3A-3). Note that the results for Pu-239, Am-241 are in general agreement with those of Romney, et al. (1976, 1978, 1985), although Price notes a discrepancy of the results with the results of Wallace for soybean plants, where Am-241 had a mean CR of 0.341. Price (1972) cited other studies of Pu and Cm uptake by plants and concludes that his results are in general agreement with the other studies. In conclusion, Price (1972) states that contamination of less than 100 pCi Np-237/g soil should be detectable in tumbleweed plants growing on the soil. The detection of Am-241, Cm-244, and especially Pu-239 by plant uptake is much less certain.

In a second study, Price (1973b) examined the effect of the application of organic acid complexes (acetate, glycolate, oxalate, and citrate) to the soil on the uptake of the same transuranic radionuclides in tumbleweed and cheatgrass. The original soil pH before addition of the complexes was 7.8. The post-treatment soil pH for the Pu, Am, and Cm organic complexes ranged from 8.1 to 9.1 m, while the pH of the Np organic acid complexes ranged from 6.5 to 7.4. Table D3-3A-4 provides the results of Price's second study.

Plant uptake of the transuranics added to the soil as organic acid complexes was in the same order as uptake from the previous study, i.e., Np > Cm ≈ Am > Pu. Differences due to chemical form were evident. Np and Pu uptake from complexes was generally greater than uptake from nitrate, whereas, uptake from Am and Cm complexes was less than from nitrate forms. Glycolate, oxalate, and citrate complexes enhanced Pu uptake by tumbleweed or cheatgrass. The enhancement with citrate was more than sixfold for tumbleweed. Am and Cm complexes of these organic acids resulted in decreased tumbleweed or cheatgrass uptake compared to nitrate forms. Am uptake by cheatgrass from the oxalate treatment was reduced by 10-fold compared to uptake from the nitrate form.

Table D3-3A-3. Mean plant uptake factors for transuranic radionuclides in tumbleweed and cheatgrass at Hanford (Price 1972).

	Np-239	Pu-239	Am-241	Cm-244
Tumbleweed (*Salsola* spp.)	1.12×10^{-1}	4.6×10^{-5}	1.40×10^{-3}	2.17×10^{-3}
Cheatgrass (*Bromus tectorum*)	1.26×10^{-2}	1.7×10^{-5}	6.0×10^{-4}	4.8×10^{-4}

Table D3-3A-4. Mean concentration ratios for four transuranic radionuclides in tumbleweed and cheatgrass at Hanford (Price 1972; 1973).

	Np-239	Pu-239	Am-241	Cm-244
Tumbleweed (*Salsola spp.*)				
Nitrate	11E-2 ± 2[a]	46E-6 ± 7	14E-4 ± 2	22E-4 ± 3
Acetate	24E-2 ± 5	48E-6 ± 4	17E-3 ± 3	12E-4 ± 1
Glycolate	23E-2 ± 7	25E-5 ± 6	21E-4 ± 4	42E-4 ± 7
Oxalate	28E-2 ± 5	27E-5 ± 5	15E-4 ± 3	15E-4 ± 3
Citrate	28E-2 ± 2	31E-5 ± 1	15E-4 ± 2	14E-4 ± 2
Cheatgrass (*Bromus tectorum*)				
Nitrate	12E-3 ± 1	17E-6 ± 2	60E-5 ± 10	48E-5 ± 5
Acetate	15E-3 ± 3	14E-6 ± 3	23E-5 ± 2	33E-5 ± 3
Glycolate	13E-3 ± 2	43E-6 ± 4	8E-5 ± 1	19E-5 ± 2
Oxalate	7E-3 ± 1	53E-6 ± 5	6E-5 ± 1	8E-5 ± 1
Citrate	11E-3 ± 2	51E-6 ± 2	10E-5 ± 2	16E-5 ± 1

a. 11E-2 ± 2 equals $11 \times 10^{-2} \pm 2 \times 10^{-2}$ (standard error of mean).

The greatest uptake was of Np-239 by tumbleweed, with CR values all above 0.2 for all organic acid complexes. The remaining radionuclides had CR values ranging from 4.8×10^{-5} (Pu-239 in acetate) to 0.017 (Am-241 in acetate) for tumbleweed. In cheatgrass, the CR values for Np-239 were between 0.007 (oxalate) and 0.015 (acetate). The remaining radionuclides had CR values ranging from 1.4×10^{-5} (Am in acetate) to 3.3×10^{-4} (Cm in acetate) for cheatgrass.

Cline (1967) studied a number of aspects of Am-241 and Pu-239 uptake by beans and barley at Hanford. He found uptake of Am-241 to be 20–30 times greater than that of Pu-239. Cline (1967) also reported that Am-241 was toxic to the roots of pea seedlings grown in 0.1 mCi A-241m/liter, but that Pu-239 did not show this toxicity, although there was restricted cell division. He concluded that "this apparent greater toxicity of Am-241 may reflect solubility differences occasioned by the approximately 50-fold greater mass of Pu-239 involved in equal-microcurie experiments. However, in view of the soil mobility data, it seems more likely that these two elements are behaving in a chemically different manner such that americium is more readily taken up and thus is able to exert a greater toxicity."

D3-3A.3.3 Los Alamos National Laboratory

Dreesen and Marple (1979) report on the uptake of Ra-226 and U from uranium mill tailings by four-wing saltbush (*Atriplex canescens*), a plant common at the NTS, from a greenhouse experiment. They found CRs of 0.031 and 0.0055 for Ra-226 and U, respectively. They also found that plants grown on soil amended with Ra-226 and U had significantly greater concentrations of these radionuclides than plants from a control soil. Dreesen and Marple conclude that Ra-226 presents a greater hazard potential than does U.

In the mid-1970s the University of California at Los Angeles Laboratory of Nuclear Medicine and Radiation Biology ran a series of experiments either directly or indirectly related to the uptake of radionuclides in plants at the NTS. The work was sponsored by the Nuclear Regulatory Commission and the project leader was A. Wallace. The only available documentation of these results is in summary form (Wallace et al., 1977), which will be discussed here.

Since heavy metals generally adhere to plant roots, an experiment was performed to see how much Am-241 would be transferred from the root to the shoots. It was found that very little Am was translocated to the shoots, though the chelator DTPA apparently enhanced the transfer of Am already accumulated in the roots to the shoots. They attributed this increase to an ability of DTPA to facilitate transport of Am across the root membrane.

Because radionuclides accumulate on roots, the degree of contamination by Am with and without DTPA in an edible root crop (radish) was studied. They found that the radish peel does contain a large amount of the radionuclide. Peeling the radish effectively removed the contamination, but washing in water and other solutions did not.

There was interest in determining if seemingly inconsequential experimental methods would have an effect on the results of the experiment. Specifically, the effect of the size of the pots in which the plants are grown and "type of application" of Am to the soil were studied. They found that the CR remained constant regardless of pot size and method of application. They also concluded that the CR remained constant regardless of the concentration of Am in the soil (apparently a fixed quantity of contaminant was added to the pots, regardless of size).

In a related study, the effect of DTPA concentration on Am uptake was studied. An increase in DTPA concentration did increase the rate of uptake at rates slightly greater than proportional to change in concentration. Hence, a doubling of the DTPA concentration more than doubled the uptake rate. It is concluded that, "this consistent result implies a very active system that increases in efficiency as the DTPA level is increased."

In the same study they found that the primary leaves (these are the first two leaves that grow on the plant, in the case of a dicot) of the bush bean contained 10 times as much Am as the trifoliate leaves. Since the primary leaves abscise before maturity, the researchers speculate that this leaf loss could be a means of eliminating Am from the plant.

In most plants, the leaves have higher CRs than the stems. However, in desert holly (*Atriplex hymenelytra*) the stem CR values were much higher than for leaves. The stem Am CRs were greater than 0.2. (Note that this is the only CR value reported in the literature for a species native to the NTS.)

Also of interest was whether plants with salt glands in their leaves would metabolically excrete Am from the leaves along with the salt. The plant Tamarix was used in the study and found that Am was not

transported out with the salts, even though the plant's salt gland is not specific for a salt and a variety of ions are pumped through it.

These researchers also suggested further studies based on previous studies. In these suggestions, they give a couple of results that were not reported previously, possibly because the results were not generated from controlled experiments. The following results may be considered tentative, but are of interest to our study.

1. Pu apparently became more available with time when clover was grown continuously in potted soil for 5 years. Whether the Pu changes or the plant transport metabolism changes is not clear. This phenomenon was also described by Newbould (1963), Newbould and Mercer (1961), and Romney, et al. (1970) for Pu.

2. Plants that actively accumulate various substances which appear unrelated to growth (and are typically detrimental to the health of other plants) are known to exist. For instance, the *Chenopodaceae* family is characterized by a number of plants that can accumulate salt. The researchers speculate that such plants may exist for transuranics. They expect that plants whose roots excrete chelators of the type that would chelate the radionuclides or plants whose roots are very high in organic acids would be the most likely candidates.

D3-3A-4. SUMMARY

It has been established that plants take up and transport to the shoot many radionuclides, including both transuranics and fission products. However, there is a large range of observed uptake rates both between and within radionuclides. Although there is a great deal of variability in the results, particularly in terms of specific radionuclide uptake rates for specific plant species, some general conclusions can be made. First and foremost is the recognition that plant uptake of radionuclides is strongly dependent on the chemistry of the soil in which the plant is growing. Soil chemistry influences the mobility of radionuclides in the soil, and therefore determines the amount of activity potentially available for uptake. Many of the factors known to increase the uptake of radionuclides are, as Price (1972) suggests, also factors that increase plant growth.

Table D3-3A-5 shows the maximum and approximate average concentration ratios of the radionuclides studied in the literature. However, some caution should be noted. First, although the literature search for this report has been fairly extensive, some studies were probably not considered that should have been. Hence, this table is not exhaustive. It is also important to recognize that concentration ratios are not always used in studies. Hence, the values reported in the table are only for studies that reported their results in CRs. Finally, not all studies reported drying temperature and time used to compute the oven-dry weights. This may cause problems in comparing results. See Price (1971) for a discussion of comparing CR. Most of the species used in finding the CRs reported in Table D3-3A-5 were annuals. Also the length of time growing in the contaminated medium differed (as did the substrate the plants were grown in), though most studies grew the plants to maturity (i.e., development of fruiting bodies). More extensive tables of plant radionuclide uptake CRs are given by Ng et al. (1982) and Grogen (1985).

Little information on plant root depths at the NTS is available. The one available study (Wallace, et al., 1980) for the NTS is not extensive, but does provide some insight on the problem. They find virtually no root growth below 50 cm for the nine native species studied. They suggest that maximum root depth corresponds to maximum water infiltration.

Table D3-3A-5. Selected range and average concentration ratios for Pu, Am, Np, U, Th, and Ra.

	Range			
Nuclide	Minimum	Maximum	Average	Reference
Am	0.002	0.003	0.00246	Cline (1967)
	0.00006	0.017	0.0028	Price (1972, 1973)
	0.00006	0.017	0.0022	Romney, et al. (1976, 1978)
	0.000082	0.0765	0.01314	Romney, et al. (1985)
	0.0000014	0.000156	0.0000624	Schulz, et al. (1976)
Am + DTPA	0.00007	0.37	0.02437	Romney, et al. (1976, 1978)
	0.00079	0.80	0.065	Romney, et al. (1985)
Pu	0.0001	0.006	0.002775	Bondietti and Sweeton (1977)
	0.00006	0.0002	0.00012	Cline (1967)
	0.00062	0.00074	0.000687	Lipton and Goldin (1976)
	0.000014	0.00031	0.00011	Price (1972, 1973)
	0.0000024	0.0011	0.000182	Romney et al. (1976, 1978)
	0.0000089	0.062	0.000723	Romney et al. (1985)
	0.0000002	0.000023	0.00000977	Schulz et al. (1976)
	0.000054	0.00056	0.0002015	Wildung and Garland (1974)
Pu + DTPA	0.458	2.48	0.888	Lipton and Goldin (1976)
	0.0000059	0.0066	0.0011	Romney et al. (1976, 1978)
	0.0000671	0.0679	0.006482	Romney et al. (1985)
Np	0.007	0.28	0.1198	Price (1972, 1973)
	0.003	0.756	0.1256	Romney et al. (1981)
	0.00003	0.853	0.0617	Romney et al. (1981)
Np + DTPA	0.002	0.971	0.139	Romney et al. (1981)
Ra	0.000021	0.6	0.036	McDowell-Boyer, et al.(1980)
	0.002	0.009	0.0058	Drichko, et al. (1984)
	0.38	2.94	1.215	Adriano, et al. (1981)
Th	0.0001	0.007	0.003275	Bondietti and Sweeton (1977)
	0.003	0.008	0.0052	Drichko, et al. (1984)
U	0.01	0.02	0.0125	Bondietti and Sweeton (1977)

The depth of water penetration would therefore be the most important factor in determining the change of plant uptake of radionuclides over the 10,000 years that must be assessed. Change in species composition could have an effect on total uptake rate, particularly if deep-rooting plants such as mesquite become established at the site. It is important to note, however, that the data of Spaulding et al. (1984) shows no evidence of mesquite over the last 10,000 years. Again, there is too little information on root depths to establish the risk.

D3-3A-5. REFERENCES

NOTE: The boldface numbers following some of the citations listed below (e.g., "**A2**" are reference numbers for the comprehensive CR summaries provided in Appendix A.

Adriano, D. C., M. S. Delaney, G. D. Hoyt, and D. Paine, 1977, "Availability to Plants and Soil Extraction of Americium-241 as Influenced by Chelating Agent, Lime, and Soil Type," *Environmental and Experimental Botany* 17:69-77.

Adriano, D. C., K. W. McLeod, and T. G. Ciravolo, 1984, "Long-Term Root Uptake of Radiocesium by Several Crops," *Journal of Plant Nutrition* 7:1415-1432. **A2**

Adriano, D. C., K. W. McLeod, and T. G. Ciravolo, 1981, "Plutonium, Curium, and Other Radionuclide Uptake by the Rice Plant from a Naturally Weathered Contaminated Soil," *Soil Science* 132:83-88. **A1**

Ananyan, V. L. and A. S. Avetisyan, 1971, "Accumulation of Radium and Calcium in Plants as Influenced by Fertilizers," *Soobshch. Inst. Agrokhim. Probl. Gidroponiki Akad. Nauk Arm. 11:36-41 (Russ.), Cf. Chemical Abstracts* 78:42117j (1973).

Arthur, W. J. III, 1982, "Radionuclide Concentrations in Vegetation at a Solid Radioactive Waste-Disposal Area in Southern Idaho," *Journal of Environmental Quality* 11:394-399. **A3**

Arthur, W. J. and O. D. Markham, 1982, "Ecological Vectors of Radionuclide Transport at a Solid Radioactive Waste Disposal Facility in Southeastern Idaho," In: *Proceedings of Canadian Nuclear Society International Conference on Radioactive Waste Management.* **A4**

Athalye, V. V., V. Ramachandran, and T. J. D'Souza, 1995, "Influence of Chelating Agents on Plant Uptake of Cr-51, Pb-210, and Po-210," *Environmental Pollution* 89:47-53. **A5**

Au, F. H. F, 1974, "The Role of Soil Microorganisms in the Movement of Plutonium," In: *The Dynamics of Plutonium in Desert Environments*, Nevada Applied Ecology Group, NVO-142.

Au, F. H. F. and W. F. Beckert, 1975, "Influence of Selected Variables on Transport of Plutonium to Spores of *Aspergillus niger*," In: *The Radioecology of Plutonium and Other Transuranics in Desert Environments*, NVO-153.

Au, F. H. F. and W. F. Beckert, 1977, *Influence of Microbial Activities on Availability and Nevada Biotransport of Plutonium*, Environmental Plutonium on the Nevada Test Site, Nevada Applied Ecology Group, Report NVO-171.

Baes, C. F, 1982, "Prediction of Radionuclide Kd Values from Soil-Plant Concentration Ratios," *Transactions of the American Nuclear Society*, 41:53-54. **B2**

Baes, C. F. and R. D. Sharp, 1981, "Predicting Radionuclide Leaching from Root Zone Soil for Assessment Applications," *Transactions of the American Nuclear Society*, 38:111-112. **B3**

Baes, C. F. III, R. D. Sharp, A. L. Sjoreen, and R. W. Shor, 1984, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*, DE-855-000287, September. **B4**

Barber, D. A., 1964, "Influence of Soil Organic Matter on the Entry of Caesium-137 into Plants," *Nature* 204:1326-1372.

Beatley, J. C., 1976, *Vascular Plants of the Nevada Test Site and Central-Southern Nevada*, Ecologic and Geographic Distributions, TID-26881.

Bernhardt, D. E. and G. G. Eadie, 1976, "Parameters for the Uptake of Transuranic Elements by Terrestrial Plants," Technical Note ORP/LV-76-2 USEPA.

Bondietti, E. A. and F. H. Sweeton, 1977, "Transuranic Speciation in the Environment," In: *Transuranics in Natural Environments*, Nevada Applied Ecology Group, Report NVO-178. **B1**

Carson, M. P. and L. J. Fraley, 1976, "Uptake of Plutonium by Cheat Grass from Contaminated Soil Cores," In: Whicker et al., 14th *Technical Progress Report on Radioecology on Natural Systems in Colorado*, COO-1156-84. **C1**

Case, C. J. Davis, R. French, and S. Raker, 1984, *Site Characterization in Connection with the Low-Level Defense Waste Management Site in Area 5 of the Nevada Test Site*, Nye County, Nevada - Final Report, DOE/NV/10162-13.

Cataldo, D. A. and R. E. Wildung, 1978, "Soil and Plant Factors Influencing the Accumulation of Heavy Metals by Plants," *Environmental Health Perspectives* 27:149-159. **C3**

Chester, R. O. and C. T. Garten Jr., 1980, "A Review of Parameters Describing Terrestrial Food-Chain Transport of Lead-210 and Radium-226," *Nuclear Safety* 21:486-494. **C4**

Cline, J. F., 1967, "Uptake of ^{241}Pu by Plants," BNWL-714. **C2**

Dahlman, R. C., E. A. Bondietti, and L. D. Eyman, 1976, "Biological Pathways and Chemical Behavior of Plutonium and Other Actinides in the Environment," In: *Actinides in the Environment, ACS Symp. Series 35*, A.M. Friedman (ed.), pp. 47-80.

Dreesen, D. R., and M. L. Marple, 1979, *Uptake of Trace Elements and Radionuclides from Uranium Mill Tailings by Four-Wing Saltbush (Atriplex canescens) and Alkali Sacaton (Sporobolus airoides)*, LA-UR-79-3045. **D1**

Drichko, V. F., and E. P. Lisachenko, 1984, "Background Concentrations of ^{226}Ra, ^{228}Th, and ^{40}K in Cultivated Soils and Agricultural Plants," *The Soviet Journal of Ecology* 15:81-85. **D2**

D'Souza, T. J. and K. B. Mistry, 1970, Comparative Uptake of Thorium-230, Radium-226, Lead-210, and Polonium-210 by Plants," *Radiation Botany*, 10:293-295. **D3**

Electric Power Research Institute (EPRI), 1990, *Soil-to-Plant Tranfer of Carbon-14 for Environmental Assessment of Radioactive Waste Repositories*, EPRI NP-6946. August. **E2**

Environmental Protection Agency (EPA), 1989, *Risk Assessment Methodology - Environmental Impact Statement (NESHAPS) for Radionuclides Background Information Document*, Volume 1, EPA/520/1-89-005. September. **E3**

Essington, E. H., E. B. Fowler, and W. L. Polzer, 1981, "The Interactions of Low-Level, Liquid Radioactive Wastes with Soils: 2. Differences in Radionuclide Distribution Amount for Surface Soils," *Soil Science* 132:13-18. **E1**

Evans, E. J. and A. J. Dekker, 1967, "The Effect of Soil Organic Matter Content on the Cs-137 Concentration in Crops," *Canadian Journal of Soil Science*, 47:7-13.

Evans, E. J. and A. J. Dekker, 1966, Plant Uptake of Cs-137 From Nine Canadian Soils," *Canadian Journal of Soil Science*, 46:167-176.

Fowler, E. B., E. H. Essington, and W. L. Polzer, 1981, "The Interactions of Low-Level, Liquid Radioactive Wastes with Soils: 1. Behavior of Radionuclides in Soil-Waste Systems," *Soil Science*, 132:2-12. **F1**

Foxx, T. S. and G. D. Tierney, 1986b, *Rooting Patterns in the Pinyon-Juniper Woodland*, LA-UR—85-3819.

Foxx, T. S., G. D. Tierney, and J. M. Williams," 1984a, *Rooting Depths of Plants on Low-Level Waste Disposal Sites*, LA-10253-MS.

Foxx, T. S., G. D. Tierney, and J.M. Williams, 1984b, Rooting Depths of Plants Relative to Biological and Environmental Factors, LA-10254-MS.

Francis, C. W., 1973, "Plutonium Mobility in Soil and Uptake in Plants: A Review," *Journal of Environmental Quality*, 2:67-70.

Gilbert, R. O., and L. L. Eberhardt, 1976, *Statistical Analysis of "A Site" Data and Interlaboratory Comparisons for the Nevada Applied Ecology Group, Studies of Environmental Plutonium and Other Transuranics in Desert Ecosystems*, Nevada Applied Ecology Group Progress, Report NVO-159. pp. 117-154.

Grogan, H. A., 1985, *Concentration Ratios for BIOPATH: Selection of the Soil-to-Plant Concentration Ratio Database*, EIR-575.

Hakonson, T. E., 1975, Environmental Pathways of Plutonium into Terrestrial Plants and Animals," *Health Physics* 29:583-588. **H1**

Hakonson, T. E., R. L. Watters, and W. C. Hanson, 1981, "The Transport of Pu in Terrestrial Ecosystems," *Health Physics* 40:63-69. **H2**

Hale, V. Q. and A. Wallace, 1970, "Effects of Chelates on Uptake of Some Heavy Metal Radionuclides from Soil by Bush Beans," *Soil Science* 109:262-263.

Harris, G. A., 1989, "Radionuclide Uptake in Plants," EGG-SARE-869. **H3**

Hawkins, D. B. and H. L. Short, 1965, *Equations for the Sorption of Cesium and Strontium on Soil and Clinoptilolite*, IDO-12046. **H4**

Hinton, T. G., M. McDonald, Y. Ivanov, N. Arkhipov, and A. Arkhipov, 1995, "Foliar Absorption of Resuspended ^{137}Cs Relative to Other Pathways of Plant Contamination," *Journal of Environmental Radioactivity*, 30:15-30.

Hoyt, G. D. and D. C. Adriano, 1976, "Americium-241 Uptake by Bahiagrass, *Paspalum notatum*, as Influenced by Lime and Organic Matter Addition and Decomposition," *Agronomic Abstracts*, p.128.

Ichikawa, R., M. Eto, and M. Abe, 1962, "Sr-90 and Cs-137 Absorbed by Rice Plants in Japan, 1960," *Science* 135:1072. **I1**

International Atomic Energy Agency (IAEA), 1994, "Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments," *Technical Reports Series No. 364*. **I2**

Jacobson, L. and R. Overstreet, 1948, "The Uptake by Plants of Plutonium and Some Products of Nuclear Fission Adsorbed on Soil Colloids," *Soil Science* 65:129-134.

Kinnear, J. E., A. Wallace, and E. M. Romney, 1981, "Frequency Distribution of Am-241 in a Population of Bush Bean Plants Grown in Soil in a Glasshouse," *Soil Science* 132:122-126. **K1**

Klepper, B., L. E. Rogers, J. D. Hedlund, R. G. Schreckhise, and K. R. Price, 1978, "Radiocesium Movement in a Gray Rabbitbrush Community," In: CONF-760429. **K2**

Klepper, B., L. E. Rogers, J. D. Hedlund, R. G. Schreckhise, and K. R. Price, 1979, *Radioactivity Associated with Biota and Soils of the 216-A-24 Crib*, Pacific Northwest Laboratory (PNL)-1948. **K3**

Lipton, W. V. and A. S. Goldin, 1976, "Some Factors Influencing the Uptake of Plutonium-239 by Pea Plants," *Health Physics* 31:425-430. **L1**

Little, C. A., T. F. Winsor, J. E. Johnson, and F. W. Whicker, 1973, *Plutonium in the Terrestrial Environs of Rocky Flats*, COO-1156-1163.

Livens, F. R., A. D. Horrill, and D. L. Singleton, 1991, "Distribution of Radiocesium in the Soil-Plant Systems of Upland Areas of Europe," *Health Physics* 60:539-545. **L2**

Lopatkina, A. P., V. S. Konarov, A. N. Sergeev, and A. G. Andreev, 1970, *Concentration of Uranium by Living and Dead Peat-Forming Plants*, Geochem. Int. (Env. transl.) 7:277-282.

Marouf, B. A., A. S. Mohamad, J. S. Taha, N. F. Wawfic, and J. A. Mahmood, 1992, "The Transfer of Cs-137 from Soil to Plants," *Environment International* 18:211-215. **M6**

Marckwordt, U, 1968, "Magnesium-Extractable Caesium in Soils and Caesium Uptake by Plant," Plant and Soil 28:372-374.

McDowell, L. M, 1976, "Physical Characterization of Plutonium in Soil," In: *Whicker et al., 14th Technical Progress Report on Radioecology of Natural Systems in Colorado*, COO-1156-84. **M1**

McDowell-Boyer, L. M., A. P. Watson, and C. C. Travis, 1980, "A Review of Parameters Describing Lead-210 and Radium-226," *Nuclear Safety* 21:486-495. **M2**

McKenzie, D. H., L. L. Cadwell, L. E. Eberhardt, W. E. Kennedy Jr., R. A. Peloquin, and M. A. Simmons, "Relevance of Biotic Pathways to the Long-Term Regulation of Nuclear Waste Disposal," *Topical Report on Reference Western Arid Low-Level Sites*, NUREG/CR-2675. **M3**

McLeod, K. W., D. C. Adriano, and T. G. Ciravolo, 1981, "Uptake of Plutonium from Soils Contaminated by a Nuclear Fuel Chemical Separations Facility," *Soil Science* 132:89-98. **M4**

Menzel, R. G, 1965, "Soil-Plant Relationships of Radioactive Elements," *Health Physics* 11:1325-1332. **M7**

Miller, M. and L. J. Fraley Jr., 1976, "Root Uptake Versus Aerial Disposition of Plutonium," In: *Whicker et al., 14th Technical Progress Report on Radioecology of Natural Systems in Colorado*, COO-1156-84. **M5**

Miner, F. J. and P. A. Glover, 1974, "Plutonium Behavior in the Soil/Water Environment in Sorption of Plutonium by Soils," *Agronomic Abstracts* p. 35.

Mistry, K. B. and B. M. Bhujbal, 1973, Effect of Calcium and Organic Matter Additions on the Uptake of Radiostrontium and Radium by Plants from Indian Soils," *Agrochemical* 18: 173-183, Cf. *Chemical Abstracts* 81:135107h (1974).

Morishima, H., T. Koga, H. Kawai, Y. Honda, and K. Katsurayama, 1976, "Experimental Studies on the Movement of Uranium in the Soil to Vegetables," *Radioisotopes* 25:773-778, Cf. *Chemical Abstracts* 87:35152z (1977).

Nagpal, M. K., K. K. Nagpaul, and A. K. Bhan, 1974, "Uranium Content in Plants: A Preliminary Report," *Curr. Sci.* (India) 43:8-9.

Newbould, P., 1963, "Absorption of Plutonium-239 by Plants," Gr. Brit. Ag. Res. Council, ARCRL 10:86.

Newbould, P. and E. R. Mercer, 1961, "Absorption of Plutonium-239 by Plants," Gr. Brit. Ag. Res. Council ARCRL 8:81-82.

Ng, Y. C., C. S. Colsher, and S. E. Thompson, 1979, "Transfer Coefficients for Terrestrial Foodchain - Their Derivation and Limitations," CONF-7810179-1. **N1**

Ng, Y. C., S. E. Thompson, and C. S. Colsher, 1982, *Soil-to-Plant Concentration Factors for Radiological Assessments*, NUREG/CR-2975.

Nishita, H., 1981, "Relative Adsorption and Plant Uptake of ^{238}Pu and ^{239}Pu in Soils," *Soil Science*. 132:66-70. **N2**

Nishita, H. and R. M. Haug, 1981, "Some Factors that Influence the Extractability of ^{239}Pu(IV) from Several Clay Minerals," *Soil Science* 132:35-39. **N3**

Nishita, H., A. Wallace, E. M. Romney, and R. K. Schulz, 1981, "Effect of Soil Type on the Extractability of ^{237}Np, ^{239}Pu, ^{241}Am, and ^{244}Cm as a Function of pH," *Soil Science* 132:25-34. **N4**

Nishita, H., A. Wallace, E. M. Romney, and J. Kinnear, 1981, "Relationship Between the Chemical Extractability of Several Transuranic Elements From Soils and Their Uptake by Wheat Plants," Soil Science 132:60-65. **N5**

Noordijk, H., K. E. van Bergeijk, J. Lambrechts, and M. J. Frissel, 1992, "Impact of Aging and Weather Conditions on Soil-to-Plant Transfer of Radiocesium and Radiostrontium," Journal of Environmental Radioactivity 15:277-286.

Nygren, P., P. Hari, T. Raunemaa, M. Kulmala, S. Luokkanen, M. Holmberg, and E. N. Nikinmaa, 1994, "Behavior of ^{137}Cs from Chernobyl Fallout in a Scots Pine Canopy in Southern Finland," Canadian Journal of Forest Research 24:1210-1215.

Odum, E. P., 1971, "Fundamentals of Ecology," 3rd Edition, W.B. Saunders Co., Philadelphia.

Otte, M. L. and A. H. B. M. Wijte, 1993, "*Environmental Variation Between Habitats and Uptake of Heavy Metals by Uritica dioica*," Environmental Monitoring and Assessment 28:263-275. **O1**

Phillips, W. S., 1963, "Depth of Roots in Soil," Ecology 44:424.

Pimpl, M. and H. Schuttelkopf, 1981, "Transport of Pu, Am, and Cm from Soils into Plants by Root Uptake," Nuclear Safety 22:214-225. **P1**

Pinder, J. E., M. H. Smith, D. Paine, K. W. McLeod, D. C. Adriano, J. C. Corey, and A. L. Boni, 1976, "Plutonium Cycling in Natural and Agricultural Ecosystems," Bulletin of the Ecological Society of America, Annual Meeting Program 57:21.

Polzer, W. L., E. B. Fowler, and E. H. Essington, 1981, "The Interactions of Low-Level, Liquid Radioactive Wastes with Soils: 3. Interaction of Waste Radionuclides with Soil from Horizons of Two Soil Series," Soil Science 132:19-24. **P2**

Price, K. R., 1971, "A Critical Review of Biological Accumulation Discrimination and Uptake of Radionuclides Important to Waste Management Practices 1943-1971," BNWL-B-148. **P3**

Price, K. R., 1972, "Uptake of ^{237}Np, ^{239}Pu, ^{241}Am and ^{244}Cm from Soil by Tumbleweed and Cheat Grass," BNWL-1688. **P4**

Price, K. R., 1973a, "Review of Transuranic Elements in Soils, Plants, and Animals," Journal of Environmental Quality 2:62-66.

Price, K. R., 1973b, "Tumbleweed and Cheat Grass Uptake of Transuranium Elements Applied to Soil as Organic Acid Complexes," BNWL-1755. **P5**

Prister, B. S. and S. S. Prister, 1970, "Effects of Uranium on the Growth and Development of Plants and its Accumulation as a Function of the Content in the Germination Medium," AEC-tr-7154; Translation of Radiobiologiva 10(l):3-160; Radiobiology 10:221-224. **P6**

Rediske, J. H., J. F. Cline, and H. H. Selders, 1955, "The Absorption of Fission Products by Plants," Hanford Atomic Products Operations, HW-36734.

Rhodes, D. W., 1957, "The Effect of pH on the Uptake of Radioactive Isotopes from Solution by a Soil," Soil Science Society of America Proceedings 1957:389-392. **R1**

Romney, E. M., E. H. Essington, E. B. Fowler, T. Tamura, and R. O. Gilbert, 1987, "Plutonium in the Desert Environment of the NTS and the Tonopah Test Range," pp. 121-130 in: Pinder et al., (Eds.), Environmental Research on Actinide Elements. CONF-841142. **R2**

Romney, E. M., H. M. Mork, and K. H. Larson, 1970, "Persistence of Plutonium in Soil, Plants, and Small Mammals," Health Physics 19:487-491. **R10**

Romney, E. M., A. Wallace, and J. E. Kinnear, 1978, "Plant Uptake of Pu and Am through Roots in NTS Soils," pp. 110-120 in: Selected Environmental Plutonium Research Reports of the NAEG. NVO-192, Vol. 1. **R3**

Romney, E. M., A. Wallace, J. E. Kinnear, and R. A. Wood, 1982 (or 1985), "Plant Root Uptake of Pu and Am. In: Radioecology of Transuranics and other Radionuclides in Desert Ecosystems," W. A. Howard, P. B. Dunaway, and R. G. Fuller (eds.), NVO-224, pp. 185-199. **R4**

Romney, E. M., A. Wallace, R. T. Mueller, J. W. Cha, and R. A. Wood, 1981, Effect of DTPA on Concentration Ratios of ^{237}Np and ^{244}Cm in Vegetative Parts of Bush Bean and Barley," Soil Science 132:104-107. **R5**

Romney, E. M., A. Wallace, R. K. Schulz, J. Kinnear, and R. A. Wood, 1981, "Plant Uptake of ^{237}Np,^{239,240}Pu, ^{241}Am, and ^{244}Cm from Soils Representing Major Food Production Areas of the United States," Soil Science 132:40-59. **R6**

Romney, E. M., A. Wallace, P. A. T. Wieland, and J. E. Kinnear, 1976, "Plant Uptake of $^{239-240}$Pu and ^{241}Am through Roots from Soils Containing Aged Fallout Materials," UCLA 12-1056. **R7**

Ronneau, C., L. Somber, C. Myttenaere, P. Andre, M. Vanhouche, and J. Cara, 1991, "Radiocesium and Potassium Behavior in Forest Trees," Journal of Environmental Radioactivity, 14:259-268.

Routson, R. C., 1975, "The Effect of Soil Concentrations on the Tumbleweed Uptake of Sr-90 and Cs-137 from a Burbank Sand," BNWL-1905. **R9**

Routson, R C., 1973, "A Review of Studies on Soil-Waste Relationships on the Hanford Reservation from 1944 to 1967," BNWL-1464. **R8**

Sawidis, T., 1988, "Uptake of Radionuclides by Plants after the Chernobyl Accident," Environmental Pollution 50:317-324. **S1**

Schulz, R. K., 1977, "Root Uptake of Transuranic Elements," pp. 321-330 in: Transuranics in Natural Environments," Nevada Applied Ecology Group, NVO-178. **S2**

Schulz, R. K. and M. R. Ruggieri, 1981, "Uptake and Translocation of Neptunium-237, Plutonium-238, Plutonium-239, 240, Americium-241, and Curium-244 by a Wheat Crop," Soil Science 132:77-82. **S4**

Schulz, R. K., E. B. Fowler, E. H. Essington, and W. L. Polzer, 1981, "Studies on Unsaturated Zone Hydrology and Radionuclide Migration at a Shallow Land Burial Site," Soil Science 132:99-103.

Schulz, R. K., R. Overstreet, and I. Barshad, 1960, "On the Soil Chemistry of Cesium 137," Soil Science 89:16-27.

Schulz, R. K., G. A. Tomplins, L. Leventhal, and K. L. Babcock, 1976, "Uptake of Pu and Am by Barley from Two Contaminated Nevada Test Site Soils," Journal of Environmental Quality 5:406-410. **S5**

Selders, A. A., 1950, "The Absorption and Translocation of Several Fission Elements by Russian Thistles," HW-18034. **S6**

Sheppard, M. I., 1985, "Radionuclide Partitioning Coefficients in Soils and Plants and their Correlation," Health Physics 49:106-111. **S7**

Sheppard, M. I., T. T. Vandergraaf, D. H. Thibault, and J. A. Keith Reid, 1983, "Technetium and Uranium: Sorptions by and Plant Uptake from Peat and Sand," Health Physics 44:635-643. **S9**

Simon, S. L. and L. J. Fraley Jr., 1986, "Ra-226 Uptake by Sagebrush from Uniform and Nonuniform Soil Amendments," Environmental and Experimental Botany 26:81-88. **S8**

Spaulding, W. G., S. W. Robinson, and R. L. Paillet, 1984, "Preliminary Assessment of Climatic Charrize Duriniz Late Wisconsin Time, Southern Great Basin and Vicinity, Arizona, California, and Nevada," U.S. Geological Survey Water-Resources Investigations Report 844328.

Sultanbaev, A. S., 1974, "Content of Natural Uranium in Soil and its Accumulation in Plants," Tr. Kirg. Nauchno-Issled. Inst. Zemled. 12:197-207 (Russ), Cf. Chemical Abstracts 84:178822s (1976).

Topcuoglu, S., A. Mine Van Dawen, and N. Gungor, 1995, "The Natural Depuration Rate of ^{137}Cs Radionuclides in a Lichen and Moss Species," Journal of Environmental Radioactivity 29:157-162.

Tierney, G. D. and T. S. Foxx, 1987, "Root Lengths of Plants on Los Alamos National Laboratory Lands," LA-10865-MS.

Van Bergeijk, K. E., H. Noordijk, J. Lembrechts, and M. J. Frissel, 1992, "Influence of pH, Soil Type, and Soil Organic Matter Content on the Soil-to-Plant Transfer of Radiocesium and Strontium as Analyzed by a Nonparametric Model," Journal of Environmental Radioactivity 15:265-276.

Vavilova, P. P. and G. V. Rusanova, 1972, "Decreasing Radium Accumulation in Agricultural Plants," Mater. Pchvam Komi ASSR Reg. Soveshch. Pochvoved. Sev.-Srednetaezhnoi Podzony Env. Chasti SSSR. 2nd 1971 (pub. 1972) 63-70 (Russ). Cf. Chemical Abstracts, 80:142495r (1974).

Wallace, A., 1971, "Effect of Soil pH and Chelating Agent (DTPA) on Uptake by and Distribution of Am-241 in Plant Parts of Bush Beans," Radiation Botany 12:433-435.

Wallace, A., 1972, "Increased Uptake of ^{241}Am by Plants Caused by the Chelating Agent DTPA," Health Physics 22:559-562. **W1**

Wallace, A. and E. M. Romney, 1972, "Radioecology and Ecophysiology of Desert Plants at the Nevada Test Site," TID-25954.

Wallace, A., E. M. Romney, and J. W. Cha, 1980, "Depth Distribution of Roots of Some Perennial Plants in the Nevada Test Site Area of the Northern Mojave Desert," Great Basin Naturalist 4:201-207.

Wallace, A., E. M. Romney, and J. E. Kinnear, 1978, "Soil, Plant, Food Chain Relationships of Transuranium Elements, Selected Annotated Bibliography," NUREG/CR-0158, UCLA 12—1135. **W2**

Wallace, A., E. M. Romney, R. T. Mueller, Sr., and S. M. Souf, 1981a, "Effect of Concentration on ^{241}Am Uptake in Plants With and Without DTPA Treatment," Soil Science 132:108-113. **W4**

Wallace, A., E. M. Romney, R. T. Mueller, Sr., and S. M. Soufi, 1981b, "Plant Uptake and Transport of ^{241}Am," Soil Science 132:114-119. **W5**

Wallace, A., R. K. Schulz, E. M. Romney, and H. Nishita, 1977, "Transuranium Element Transport in Agricultural Systems (Soils to Food Chain Transfer of Nuclear Fuel Cycle Radionuclides)," Annual Progress Report. UCLA 12-1134.

Wallace, A., E. M. Romney, and R. T. Mueller, Sr., 1981, "Effect of Various Concentrations DTPA Chelating Agent in Soil on Uptake and Distribution of Am-241 in Bush Bean Plants," Soil Science 132:120-121. **W3**

Wheeler, H. and P. Hanche, 1971, "Pinocytosis and Membrane Dilation in Uranyl-Treated Plant Roots," Science 966: 68-71.

Whittaker, R. H., 1975, "Communities and Ecosystems," 2nd Edition, Macmillan Publ. Co., N.Y.

Wilding, M. W. and D. W. Rhodes, 1963, "Removal of Radioisotopes from Solution by Earth Materials from Eastern Idaho," IDO-14624. **W7**

Wildung, R. E. and T. R. Garland, 1974, "Influence of Soil Plutonium Concentration on Plutonium Uptake and Distribution in Shoots and Roots of Barley," Journal of Agricultural and Food Chemistry 22:836-838. **W6**

Wilson, D. O. and J. F. Cline, 1966, "Removal of Plutonium-239, Tungsten-185, and Lead-210 from Soils," Nature 209:941-942.

Appendix D3
Attachment 3B

Summary of Plant Uptake Factors (PUFs) for Cs-137, Sr-90, and Tc-99

CONTENTS

TABLES

Appendix D3
Attachment 3B

Summary of Plant Uptake Factors (PUFs) for Cs-137, Sr-90, and Tc-99

D3-3B-1. INTRODUCTION

Plant uptake represents a potentially significant pathway by which radionuclides in the soil may enter the food chain. A commonly used measure of plant uptake is the *Concentration Ratio* (CR), alternatively known as *Concentration Factor* (CF) or *Plant Uptake Factor* (PUF). Regardless of which term is used, this is defined as the ratio of the radionuclide activity in the plant material to the activity in the soil within the rooting zone, or:

$$CR = CF = PUF = \frac{Radionuclide\ activity\ in\ plant}{Radionulcide\ activity\ in\ soil} = \frac{Ci/g\ oven\ dry\ vegetation}{Ci/g\ oven\ dry\ soil}$$

In most cases, CRs are less than 1.0 indicating that concentrations in plant tissue are less than those in the rooting zone of the soil supporting the plant. CR values less than 1.0 indicate that the plant does not actively assimilate the radionuclides, whereas CRs of greater than 1.0 indicate that either the plant actively absorbs the radionuclides or stores the nuclides after absorption.

Concentration ratios are typically based only on the shoot (leaf and stem) portion of the plant, and not on the root. Root CRs are typically far higher than those of the above-ground portions of the plantXan important factor when considering root crops.

Published CR values exhibit substantial variability both between different radionuclides and different plant species. Variability in CR values is also associated with factors related to the chemistry and physical properties of the soil, the chemical form of the radionuclide, meteorological conditions, and other intrinsic and extrinsic characteristics. Many of these variables that influence plant uptake are related. For example, the soil type influences the plant species that can grow on it, and the chemical form of the radionuclide, and the relative amount of radioactivity available for plant uptake. The high inherent variability of reported CR values substantially complicates the modeling of radionuclide migration within the biosphere.

The purpose of this report is to summarize plant uptake values reported for three radionuclides of concern in the management of low-level radioactive waste: Cs-137, Sr-90, and Tc-99. A fourth important radionuclide, C-14, is not discussed because plants take up carbon from the atmosphere rather than through the roots, thereby rendering the definition of CR meaningless for this element (Baes et al. (1984). Root respiration, litterfall, and senescence of fine roots results in a net transfer of carbon from the plant to the soil rather than the other way around. Elevated activities of C-14 in plants arise from the fixation of atmospheric CO_2-14.

Considerable work has been done with Cs-137 and Sr-90 both during atmospheric nuclear testing in the 1960s and post-Chernobyl in the late 1980s to the present. Less information is available on Tc-99. The assumption here is that isotopic effects are insignificant for Cs, Sr, and Tc. That is to say that Cs-134

and Cs-137 behave identically in terms of plant uptake. There is some evidence that this is not the case, but most evidence confirms that differences are minimal for these elements.

In general, the alkali metals of Group IA (including Cs) and the alkaline earth metals of Group IIA (including Sr) are relatively easily taken up from the soil by plants (Baes et al. 1984). Many of the lighter of these elements are essential plant nutrients. These elements may be actively or passively taken up and maintained by plants, potentially accumulating to higher concentrations in the plant than in the soil (i.e. CR > 1).

Both Cs and Sr are important from an ecological perspective because they are chemically similar to certain essential plant nutrients. Factors responsible for preferential uptake of these plant nutrients may also allow for the preferential uptake of the chemical analogs. Similarly, physiological or biochemical processes that act to concentrate chemical elements in various plant (or animal) tissues may also tend to accumulate analogous radionuclides.

The bioavailability of Cs and Sr decreases in most agricultural areas with time due to aging effects (i.e. irreversible adsorption and incorporation of nuclides into the soil mineral lattice). For Sr-90, this decrease will be about 3% per year, and for Cs-137 about 15% per year during the first two years following contamination (IAEA, 1994). This rate of reduction decreases with time.

D3-3B-1.1 Strontium

Sr is one of the best studied of all elements in the periodic table with respect to plant uptake (Baes et al. 1984). Sr (as well as Ra) is a Group IIA element, and is therefore chemically analogous to Ca and Mg. Considerable evidence exists that indicates that Sr is substituted for Ca in terms of both uptake and assimilation. Coughtrey and Thorne (1983) also indicated that the Ca content of a soil may considerably influence the transfer of Sr. Therefore, Sr winds up in the cell walls of plants, where most Ca is found. Uptake of Sr-90 may therefore increase in Ca deficient soils.

Published values for uptake of Sr range from about 0.077 to 17, and follow a lognormal distribution (Baes et al. 1984). The geometric range reported by Baes et al. (1984) was 2.7 for Sr.

Regarding uptake of Sr, Lakanen and Paasikallio (1970) reported a reduction in uptake of Sr when the soil organic matter content decreased. The interaction of Sr^{+2} with clay minerals is weaker than for Cs^{+}, whereas the association of Sr with humic or fulvic acids in the organic fraction is much stronger. This stronger association may be the consequence of chelationXcomplexation of the divalent Sr ions at adjacent anionic sites of a humic or fulvic acid (Saar and Weber, 1982; Sanchez et al. 1988). Sr bound in this way is not readily available to plants.

One of the few available studies on non-crop species was conducted by Routson (1975) on *Salsola kali* (tumbleweed or Russian thistle), an introduced weed well adapted to an arid climate that readily invades disturbed sites. This species has been shown to be capable of concentrating fission products, particularly Sr-90. Routson examined uptake of Sr and Cs at fairly low soil concentrations in a calcareous, sandy soil with low CEC. He found that CF was constant over several orders of magnitude of soil concentration.

D3-3B-1.2 Cesium

As with Sr, considerable information is available with respect to plant uptake of Cs, especially for crop species. Cs is chemically analogous to K, and is readily substituted for K in terms of both uptake and assimilation. Baes et al. (1984) examined 18 references for plant uptake of Cs, and found a lognormal distribution ranging from 0.018 to 0.52. The geometric mean was reported as 0.08 for these references. In comparison with Sr, the more narrow range of values reported for Cs is likely due to smaller number of observations available for Cs.

Because it is analogous to K, the uptake of Cs (including Cs-137) may be enhanced in K-deficient soils. Nygren et al. (1994) concluded that Cs from the Chernobyl accident had over time become mixed with the chemically analogous K, and was being recirculated within the trees together with K. Other factors influencing Cs-137 uptake include the following:

- **K content of soil**—Uptake of Cs has been found to be influenced by the K content of the soil and by the addition of K to the soil (e.g. Evans and Dekker, 1966; Coughtrey and Thorne, 1983). Over the long term, Cs uptake may by impacted by the depletion of soil reserves of K due to harvesting. Although the exchangeable K content of the soil is an important factor in the uptake of Cs-137, the International Union of Radioecologists (IUR) recently concluded that there is presently insufficient information on this factor to allow for statistical analyses (IUR, 1992).

- **Soil content and type of clay**—The higher transfer of Cs in sand compared to loam or clay is probably largely explained by differences in the clay content between soils, although the composition of the clay minerals may be important as well (Nielsen and Strandberg, 1988). Schulz et al. 1960 found significant differences in Cs uptake depending on the clay minerals present (e.g. montmorillonite, kaolinite, illite, vermiculite, and chlorite). Cs is much more efficiently retained than K by clay soils and by organic matter in forest litter. Consequently, in a forest system, there is a progressive transfer of atmospherically deposited Cs from trees to the soil.

- **Organic matter content of soil**—Barber (1964) found significant correlation between Cs-137 uptake by perennial ryegrass and percentage organic matter in soils where high organic matter contents were associated with high Cs-137 concentration in the plant. The effect on transfer of Cs may be explained by the large cation exchange capacity (CEC) of organic matter and the spatial distribution of organic substances around clay particles, which prevents adsorption and subsequent fixation of Cs on the clay minerals (e.g., Fredriksson et al. 1966). The interaction of Cs^+ with organic substances is weak, so that the ions remain available to plants (Saar and Weber, 1982; Sanchez et al. 1988). Others have found correlations between Cs uptake and K, NH_4^+, and other cations in the soil. Under conditions of high soil organic matter contents in combination with high soil moisture content and low temperatures, microorganisms produce NH_4^+ ions which prevent fixation of Cs-137 and thus counteract the decrease of the bioavailability.

- **Irrigation**—The IUR (and others) have recognized the influence of irrigation on radionuclide uptake (IUR 1992).

- **Litter layer**—The presence of a litter layer to which Cs is adsorbed and thus remains more available with time.

- **Fertilization**—Seel et al. (1995) observed a dramatic increase in effect of fertilization with K on the uptake of Cs-137 in vegetables, with concentrations in plants receiving the K being less than half of the concentrations in plants that did not receive K. Significant differences among species and plant parts were also observed.

- **Plant species**—The IUR working group on Soil-to-Plant Transfer Factors noted that the composition of the vegetation within a system will have a large impact on plant uptake in a community basis (IUR, 1992). For example, plants like clover (Trifolium, Medicago) and Ranunculus have much larger CRs than grass.

D3-3B-1.3 Technetium

Among the Period V transition elements, CRs for Tc are fairly well documented (Baes et al. 1984). Sheppard et al. (1983) found Tc-99 uptake by Swiss chard was four orders of magnitude higher in a sandy soil than in a peat soil. When soil fixation occurs, as in the peat soil, this sorption becomes the controlling factor in the plant uptake of Tc.

D3-3B-2. PROBLEMS WITH EARLY CONCENTRATION RATIO STUDIES

The problems with the early concentration ratio studies are as follows:

- Many were fallout studies where the ground surface was contaminated. This presents several problems with interpretation. For example, the definition of CR involves use of the "rooting volume of soil" to determine soil concentrations. However, only a small fraction of the soil is actually contaminated. Rooting habit and sampling technique will have substantial influence on the calculated soil concentration. Also, many of the studies involving surficial contamination failed to consider resuspension. However, Hinton et al. (1996) indicated that although foliar absorption of Cs from suspended soil is measurable, it is generally inconsequential relative to other plant contamination pathways. This would imply that washing of plants prior to analysis should be done. Finally, if Cs and/or Sr remain in the upper few mm of a mineral soil, it is generally because they have become bound by the soil particles. Subsequent transport of these contaminants is subsequently governed by the physical movement of the soil to which the contaminant is sorbed.

- Many early studies of Tc uptake used high soil concentrations of pertechnetate anion (TcO_4^-), reporting uptake factors ranging from 100 to 1,000. Hoffman et al. (1980) indicated that the results of these studies were misleading in most cases because of the high Tc concentrations added to the soils and the measurement of CRs before plant maturity. Furthermore, Tc in soil becomes increasingly sorbed and therefore less available for plant uptake with time (Gast et al. 1979; Landa et al. 1977). Aging of soils over 100 days decreased observed CRs by factors of 1.5 to 5.1 in one study (Cataldo, 1979). Application of short-term pot studies to long-term assessments is clearly inappropriate for Tc. Therefore, the CR representing field measurements of long-term Tc uptake in plants reported by Hoffman represent the best estimates, according to Baes et al. 1984.

- Concentration ratios listed by the Nuclear Regulatory Commission in Regulatory Guide 1.109 (USNRC 1977) represent the ratios of the concentrations in *wet* vegetation to that of *dry* soil. Others (e.g. Ng et al. 1979; Marouf et al. 1992) reported CRs in this manner as well. Dry weight may only account for from 5 to 30% of the wet weight for vegetables, or 55 to 85% of the wet weight for grains (IAEA 1994).

- Little information is available regarding uptake of nuclides by native plants (especially in arid systems), and even less in undisturbed areas. For example, Arthur (1982) reported concentrations of radionuclides (including Cs-137 and Sr-90) in crested wheatgrass (*Agropyron cristatum*) and Russian thistle (*Salsola kali*) at the INEEL (at the Radioactive Waste Management Complex [RWMC]). However, both these two species are exotics, and the soils were disturbed.

- CR in a given crop varies in a complex manner with soil texture, and other soil properties such as CEC, exchangeable Ca, exchangeable K, pH, and organic matter content. Van Bergeijk et al. (1992) noted that transfer of Cs from soil to plants increased with increasing organic matter content, while Sr decreased with increasing organic matter content. CR also varies with crop variety, stage of growth, and plant part, as well as with experimental conditions such as the manner in which the isotope was introduced into the soil. Some radionuclides may concentrate in edible parts of grains, vegetables, fruits, and

forage plants cultivated under different conditions. Clay fraction, type of clay minerals, organic matter content, pH, soil moisture content, and amount of soluble and exchangeable Ca, K. and NH_4^+ are regarded as the main soil factors influencing transfer (e.g. Russell, 1966; Nielsen and Strandberg, 1988). Soil to plant transfer is also affected by plant parameters such as growth and development, species, and variety (Russell and Newbould, 1966; Evans and Dekker, 1968). Soil properties as well as plant parameters may interact in various ways (Coughtrey and Thorne, 1983). Soil pH hardly had any effect on the transfer of Cs in a pH range of 3.9 to 8.4,. which is in agreement with results cited by Fredriksson et al. (1966). A generally lower transfer rate of Sr at higher pH is also mentioned by Coughtrey and Thorne (1983).

- As defined by Ng et al. (1979), CF represents "the ratio of the concentration in dry plant matter to that in the soil *under steady-state or equilibrium conditions*." However, establishing and maintaining steady state conditions was not considered in most studies, especially laboratory and/or agricultural studies (e.g., Cataldo, 1979).

D3-3B-3. SUMMARIES

Concentration ratios for Sr-90, Cs-137, and Tc-99 as summarized in the literature, are provided in a series of tables below. It should be cautioned, however, that virtually all of the values provided are for crop species in non-arid regions.

Table D3-3B-1 contains ranges of values for these three radionuclides as provided by various publications and provide suggestions for use in INEEL risk assessments. Table D3-3B-2 contains information from Ng et al. (1979) on Sr-90 and Cs-137 CRs for crops grown in the Savannah River area. Values for similar crops grown in a different soil type in Denmark are also provided for comparison purposes.

Calculated concentration ratio values for various crops published by the International Atomic Energy Agency (IAEA) are provided in Table D3-3B-3 for Sr-90, Tc-99, and Cs-137 (IAEA 1994). These values are listed not only by crop type, but also by soil type and soil pH.

D3-3B-4. INTERNATIONAL UNION OF RADIOECOLOGY

The IUR has a working group that is focused on soil-to-plant transfer factors defines the concentration ratio (as the ratio of the activity per unit dry weight of plant or plant part to the activity per unit dry weight of the soil. Predictions are based on the radionuclide concentration in the soil in a 10 cm layer for grass and a 20 cm layer for other crops.

In a report of the working group in 1992, the IUR provided separate values for three gross soil categories (clay, sand, and peat) for a variety of crop species (IUR, 1992). This summary likely provides the best collection of concentration ratio information for crop species, at least for Cs-137 and Sr-90. This working group of the IUR publishes best estimates of transfer factors periodically. A summary of the numbers they have produced over the years is found in Table D3-3B-4 for Sr-90 and Table D3-3B-5 for Cs-137. Based on these numbers, the IUR also provided recommended concentration ratios for these two nuclides (Tables D3-3B-6 and D3-3B-7) for specific crops.

Table D3-3B-1. Published concentration ratio ranges for Cs-137, Sr-90, and Tc-99 and recommended values for use at INEEL.

	Cs-137	Sr-90	Tc-99
Menzel, 1965	1.0E-02 to 1.0E+00	1.0E+00 to 1.0E+02	—
Routson, 1975	5.3E-02 (Russian thistle)	1.9E+01 (Russian thistle)	—
Ng et al. 1979	8.9E-02 (3.8E-03 to 5.7E-01)	3.5E+00 (1.2E-01 to 2.3E+01)	—
Baes et al. (1984)	8.0E-02	2.5E+00	9.5E+00
IAEA, 1994	5.0E-03 to 5.3E-01 (grass)	2.0E-02 (1.7E+00 fodder) to 3.0E+00 (green veg.)	7.3E-01 (grain), 7.6E+01 (grass) and 2.6E+03 (spinach)
Harris, 1989	1.0E-02 to 1.0E+00	1.0E+00 to 1.00E+02	1.0E+00 to 1.0E+02
Marouf et al. 1992	1.0E-02 to 1.0E-01[a]	—	—
EPA, 1989	8.0E-02	2.5E+00	9.5E+00
Suggested for Crops	4.6E-01 (mixed greens IAEA, 1994)	3.0E+00 (IAEA, 1994)	2.6E+03 (IAEA, 1994)
Suggested for INEEL native plants	5.3E-01 (EPA, 1989)	1.9E+01 (Routson, 1975)	7.6E+01 (IAEA, 1994)

a. Based on plant fresh weight rather than dry weight.

Table D3-3B-2. Calculated concentration ratios[a] for crops grown in the Savanah River region (from Ng et al. 1979).

	Southeastern U.S. Sandy Loam Soil		Denmark Clay Loam Soil	
Crop	Sr-90	Cs-137	Sr-90	Cs-137
Corn	0.034	0.026	—	—
Soybeans	0.71	0.089	—	—
Wheat	0.27	0.045	0.12	<0.01
Oats	0.27	0.045	0.14	<0.01
Barley	0.27	0.045	0.12	<0.01
Rye	0.27	0.071	0.090	<0.01
Apples	0.032	0.019	0.011	0.021
Tomatoes	0.024	0.0072	—	—
Cabbage	0.08	0.004	0.058	0.01
Sweet Corn	0.011	0.0081	—	—
Snap beans	0.03	0.005	0.096	0.0066
Irish potatoes	0.06	0.02	0.014	0.0032
Hay	0.72	0.14	—	—

a. Note that concentration ratios in this study were calculated using activities in *wet* vegetation rather than dry weight. This was done to enable comparisons to be made with values published in Reg Guide 1.109 (USNRC 1979).

Table D3-3B-3. Soil-to-plant transfer factors (i.e., concentration ratios) based on activity per *dry* wt. crop per unit activity per dry weight soil (IAEA 1994).

	Crop	Expected	95% Confidence Range		Soil Type	pH	Ref
Sr-90	Cereals	1.2E-01	2.2E-02	6.6E-01	Clay, loam	6	a
	Cereals	2.1E-01	3.2E-02	1.4E+00	Sand	5	a
	Cereals	2.0E-02	2.0E-03	2.0E-01	Peat	4	a
	Fodder	1.9E-01	1.9E-02	1.9E+00	Clay, loam	6	a
	Fodder	1.0E+00	1.0E-01	1.0E+01	Sand	5	a
	Fruit	2.0E-01	5.0E-02	8.0E-01	Sand	5	a
	Grass	1.1E+00	4.0E-01	2.9E+00	Clay, loam	6	a
	Grass	1.7E+00	3.5E-01	7.8E+00	Sand	5	a
	Grass	3.4E-01	3.4E-02	3.4E+00	Peat	4	a
	Pea, bean	1.3E+00	3.4E-01	4.9E+00	Clay, loam	6	a
	Pea, bean	2.2E+00	5.3E-01	9.4E+00	Sand	5	a
	Root crops	1.1E+00	1.1E-01	1.1E+01	Clay, loam	6	a
	Root crops	1.4E+00	1.4E-01	1.4E+01	Sand	5	a
	Tubers (potato)	1.5E-01	1.8E-02	1.3E+00	Clay, loam	6	a
	Tubers (potato)	2.6E-01	5.0E-02	1.4E+00	Sand	5	a
	Tubers (potato)	2.0E-02	2.0E-03	2.0E-01	Peat	4	a
	Green vegetables	2.7E+00	7.4E-01	1.0E+01	Clay, loam	6	a
	Green vegetables	3.0E+00	3.0E-01	3.0E+01	Sand	5	a
	Green vegetables	2.6E-01	2.6E-02	2.6E+00	Peat	4	a
	Hop	8.0E-01			None	None	b
Tc-99	Cereals	7.3E-01	7.3E-02	3.7E+00	None	None	b
	Fodder	8.1E+00	8.1E-01	8.1E+01	None	None	b
	Grass	7.6E+01	1.0E+01	7.6E+02	None	None	b
	Pea, bean	4.3E+00	1.0E+01	4.3E+01	None	None	b
	Turnip	7.9E+01			None	None	b
	Potato	2.4E-01	2.4E+00	2.4E+00	None	None	b
	Cabbage	1.2E+01	1.0E+01	1.2E+02	None	None	b
	Lettuce	2.0E+01	1.0E+01	2.0E+03	None	None	b
	Spinach	2.6E-03	2.6E+02	7.8E+03	None	None	b
Cs-137	Cereals	1.0E-02	1.0E-03	1.0E-01	Clay, loam	6	a
	Cereals	2.6E-02	2.6E-03	2.6E-01	Sand	5	a

Table D3-3B-3. (continued).

Crop	Expected	95% Confidence Range		Soil Type	pH	Ref
Cereals	8.3E-02	8.3E-03	8.3E-01	Peat	4	[a]
Fodder	1.7E-02	1.7E-03	1.7E-01	Clay, loam	6	[a]
Fodder	2.9E-01	2.9E-02	2.9E+00	Sand	5	[a]
Fodder	3.0E-01	3.0E-02	3.0E+00	Peat	4	[a]
Grass	1.1E-01	1.1E-02	1.1E+00	Clay, loam	6	[a]
Grass	2.4E-01	2.4E-02	2.4E+00	Sand	5	[a]
Grass	5.3E-01	5.3E-02	5.3E+00	Peat	4	[a]
Pea, bean	1.7E-02	2.1E-03	1.4E-01	Clay, loam	6	[a]
Pea, bean	9.4E-02	1.2E-02	7.5E-01	Sand	5	[a]
Root crops	4.0E-02	4.0E-03	4.0E-01	Clay, loam	6	[a]
Root crops	1.1E-02	1.1E-03	1.1E-01	Sand	5	[a]
Tubers (potato)	7.0E-02	7.0E-03	7.0E-01	Clay, loam	6	[a]
Tubers (potato)	1.7E-01	1.7E-02	1.7E+00	Sand	5	[a]
Tubers (potato)	2.7E-01	2.7E-02	2.7E+00	Peat	4	[a]
Green vegetables	1.8E-01	1.9E-02	1.7E+00	Clay, loam	6	[a]
Green vegetables	4.6E-01	4.7E-02	4.5E+00	Sand	5	[a]
Green vegetables	2.6E-01	2.5E-02	2.7E+00	Peat	4	[a]
Rice	5.0E-03			None	None	[c]
Tomato fruit	2.2E-01			None	None	[c]

a. Frissel et al. 1992

b. Frissel and Bergeijk 1989

c. Myttenaere et al. 1969.

Table D3-3B-4. Transfer factors for Sr-90 published by the International Union of Radioecologists from 1984 through 1992 for various crop types (from IUR 1992). All values are for soils with pH = 6.

	pH	Grass	Cereals	Potato	Pods	Green Vegs.	Fodder	Root Crop
IUR84								
Clay	6	0.42	0.048	0.07	0.22	1.1	0.37	0.12
Sandy	6	0.37	0.041	0.055	0.18	0.92	0.30	0.10
IUR85								
Clay	6	1.2	0.13	0.066	0.22	1.1	0.81	0.12
Sandy	6	0.86	0.09	—	0.18	0.92	0.56	0.26
IUR87								
Clay	6	0.26	0.17	0.020	1.1	3.2	0.85	1.2
Sandy	6	0.19	0.11	0.13	0.73	2.2	0.58	0.82
IUR89								
Clay	6	1.5	0.37	0.20	0.52	0.61	0.59	0.50
Sandy	6	2.3	0.59	0.33	0.84	0.97	0.93	0.61
IUR92								
Clay	6	1.1	0.12	0.15	1.3	2.8	0.79	1.1
Sandy	6	1.7	0.21	0.26	2.2	3.1	1.0	1.4
Peaty	6	0.34	0.02	0.02	—	0.28	—	—

Table D3-3B-5. Transfer factors for Cs-137 published by the International Union of Radioecologists from 1984 through 1992 for various crop types (from IUR 1992). All values are for soils with pH=6.

	pH	Grass	Cereals	Potato	Pods	Green Vegs.	Fodder	Root Crop
IUR84								
Clay	6	0.043	0.006	0.014	0.025	0.048	0.37	0.12
Sandy	6	0.088	0.012	0.028	0.050	0.089	0.30	0.10
IUR85								
Clay	6	0.047	0.011	0.014	0.025	0.048	0.81	0.12
Sandy	6	0.041	0.016	0.028	0.050	0.089	0.56	0.26
IUR87								
Clay	6	0.13	0.015	0.074	0.084	0.19	0.85	1.2
Sandy	6	0.15	0.018	0.089	0.10	0.22	0.58	0.82
IUR89								
Clay	6	0.17	0.055	0.20	0.068	0.042	0.59	0.50
Sandy	6	0.19	0.061	0.22	0.076	0.66	0.93	0.61
IUR92								
Clay	6	0.10	0.010	0.07	0.017	2.8	0.79	1.1
Sandy	6	0.24	0.026	0.27	0.094	3.1	1.0	1.4
Peaty	6	0.53	0.083	0.17	—	0.28	—	—

Table D3-3B-6. Recommended soil to plant transfer factors for Sr-90 (IUR 1992).

	N	Recommended Value	Soil Type	pH
Cereals, grain	81	0.12	Clay, loam	6
	81	0.21	Sand	5
	4	0.020	Peat	4
Fodder	36	0.79	Clay, loam	6
	50	1.0	Sand	5
Fruit	12	0.20	Sand	5
Grass	70	1.1	Clay, loam	6
	115	1.7	Sand	5
	4	0.34	Peat	4
Pea, bean–Pod	95	1.3	Clay, loam	6
	56	2.2	Sand	5
Root crops	11	1.1	Clay, loam	6
	23	1.4	Sand	5
Tubers (potato)	29	0.15	Clay, loam	6
	113	0.26	Sand	5
	2	0.02	Peat	4
Green vegetable	65	2.7	Clay, loam	6
(except spinach)	49	3.0	Sand	5
	2	0.26	Peat	4
Hop	1	0.80	NA	NA

Table D3-3B-7. Recommended soil to plant transfer factors for Cs-137 (IUR 1992).

	N	Recommended Value	Soil Type	pH
Cereals, grains	220	0.01	Clay, loam	6
	132	0.026	Sand	5
	14	0.083	Peat	4
Fodder	173	0.017	Clay, loam	6
	22	0.29	Sand	5
	2	0.30	Peat	4
Grass	246	0.11	Clay, loam	6
	229	0.24	Sand	5
	21	0.53	Peat	4
Pea, beanXPod	124	0.017	Clay, loam	6
	63	0.094	Sand	5
Root crops	18	0.040	Clay, loam	6
	17	0.011	Sand	5
Tubers (potato)	67	0.070	Clay, loam	6
	79	0.17	Sand	5
	3	0.26	Peat	4
Mixed green vegetables	165	0.18	Clay, loam	6
(except spinach)	90	0.46	Sand	5
	2	0.26	Peat	4
Rice (irr.) soil to plant	—	0.0050	NA	NA
Rice (irr.) water to plant	—	20.	NA	NA
Tomato fruit	2	0.22	NA	NA

The IUR working group also provided corrections in concentration ratios for pH (Table D3-3B-8, normalized to pH = 6) and for post-contamination time lag (Table D3-3B-9, normalized to 0 years). Although these correction factors were adapted for crop species, they may be equally applied to non-crop species.

Table D3-3B-8. IUR concentration ratio correction factors for pH effects for Cs-137 and Sr-90 (correction relative to pH = 6).

	Cs-137				Sr-90			
pH->	4	5	6	7	4	5	6	7
IUR84	2.96	1.72	1	0.85	1.04	1.02	1	0.98
IUR85								
Cereal, fodder, grass	5.67	2.38	1	0.42	1.02	1.01	1	0.99
Pods, tubers, vegetables	2.96	1.72	1	0.58	1.43	1.20	1	0.84
IUR87	3.58	1.90	1	0.53	1.12	1.06	1	0.95
IUR89	1.83	1.35	1	0.74	2.30	1.52	1	0.66
IUR90								
Grass	1.64	1.28	1	0.78	1.32	1.15	1	0.87
Cereals	4.13	2.03	1	0.49	1.27	1.13	1	0.89
IUR92	1.69	1.30	1	0.77	1.43	1.20	1	0.84

Table D3-3B-9. Concentration ratio correction factors for time lag effects for Cs-137 and Sr-90 (correction relative to a time lag of 0 years.

	Cs-137					Sr-90				
Ph->	0	1	2	3	4	0	1	2	3	4
IUR84	1	0.90	0.82	0.74	0.60	1	0.98	0.96	0.95	0.91
IUR85										
Cereal, fodder, grass	1	0.97	0.95	0.92	0.87	1	0.99	0.99	0.98	0.97
Pods, tubers, vegetables	1	0.90	0.82	0.74	0.61	1	0.84	0.70	0.58	0.41
IUR87	1	0.92	0.85	0.78	0.66	1	0.95	0.90	0.85	0.77
IUR89	1	0.88	0.77	0.68	0.53	1	0.97	0.94	0.91	0.86
IUR90										
Grass	1	0.82	0.67	0.55	0.37	1	0.96	0.92	0.87	0.82
Cereals	1	0.69	0.48	0.33	0.16	1	0.98	0.96	0.94	0.90
IUR92	1	0.84	0.70	0.59	0.42	1	0.97	0.94	0.91	0.86

D3-3B-5. REFERENCES

Arthur, W. J. III, 1982, "Radionuclide Concentrations in Vegetation at a Solid Radioactive Waste-Disposal Area in Southern Idaho," *Journal of Environmental Quality*, 11:394–399.

Baes, C.F. III, R. D. Sharp, A.L. Sojourn, and R.W. Shore, 1984, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture* DE-855-000287, September.

Barber, D. A., 1964, "Influence of Soil Organic Matter on the Entry of Caesium-137 into Plants," *Nature* 204:1326-1372.

Cataldo, D. A., 1979, *Behavior of Technetium-99 and Iodine in a Hanford Sand and Associated Subsoil: Influence of Soil Aging on Uptake by Cheatgrass and Tumbleweed*, Pacific Northwest Laboratory, report no. PNL-2740, March.

Coughtrey, P.J. and M.C. Thorne, 1983, *Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems; a Critical Review of Data.*, Volume 1. A. A. Balkema, Rotterdam, 496 pp.

Environmental Protection Agency (EPA), 1989, *Risk Assessment Methodology, Environmental Impact Statement (NESHAPS) for Radionuclides Background Information Document*, Volume 1, EPA/520/1-89-005, September.

Evans, E.J. and A.J. Dekker, 1967, "The Effect of Soil Organic Matter Content on the Cs-137 Concentration in Crops," *Canadian Journal of Soil Science*, 47:7-13.

Evans, E. J. and A.J. Dekker, 1966, "Plant Uptake of Cs-137 from Nine Canadian Soils," *Canadian Journal of Soil Science*, 46:167-176.

Fredriksson, L., R. J. Garner, and R. S. Russell, 1966, "Cesium-137," In: *Radioactivity and the Human Diet*, ed. R. S. Russell, Pergamon Press, Oxford, UK., pp. 317-52.

Frissel, M. J., 1992, "An Update of the Recommended Soil-to-Plant Transfer Factors," *Eighth Report of IUR Working Group on Soil-to-Plant Transfer Factors*, IUR, Buren, Belgium.

Frissel, M. J. and K. E. Bergeijk, 1989, "Mean Transfer Values Derived by Simple Statistical Regression Analysis, *Sixth Report of IUR Working Group on Soil-to-Plant Transfer Factors*, RIVM, Bilthoven.

Gast, R. G., E. R. Landa, L. J. Thorvig, D. F. Grigal, and J. C. Balogh, 1979, *The behavior of Technetium-99 in Soils and Plant*: Final Report for the Period April 1, 1974 to December 31, 1978, COO-2447-6, National Technical Information Service, Springfield, Virginia.

Harris, G. A., 1989, *Radionuclide Uptake in Plants*, EGG-SARE-869.

Hinton, T. G., M. McDonald, Y. Ivanov, N. Arkhipov, and A. Arkhipov, 1995, "Foliar absorption of resuspended ^{137}Cs relative to other pathways of plant contamination," *Journal of Environmental Radioactivity*, 30:15-30.

Hoffman, F. O., J. W. Huckabee, D. M. Lucas, C. Y. Gargen Jr., T. G. Scott, R. L. Walker, P. S. Gouge, and C. V. Holmes, 1980, *Sampling of Technitium-99 in Vegetation and Soils in the Vicinity of Operating Gaseous Diffusion Facilities*, ORNL/TM-7386.

International Atomic Energy Agency (IAEA), 1994, ","Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments," *Technical Reports Series*, No. 364.

International Union of Radioecology (IUR), 1992, VIIIth report of the working group - Soil-to-plant transfer factors, *Report of the Working Group Meeting in Madrid, Spain, June 1-3, 1992*, IUR Publication, R-9212-02.

Lakanen and Paasikallio, 1970.

Landa, E. R., L. H. Thorvig, and R. G. Gast, 1977, "Effect of Selective Dissolution, Electrolytes, Aeration, and Sterilization on Technetium-99 Sorption by Soils," *Journal of Environmental Quality*, 6:181-187.

Menzel, R. G., 1965, "Soil-Plant Relationships of Radioactive Elements," *Health Physics*, 11:1325-1332.

Marouf, B. A., A. S. Mohamad, J. S. Taha, N. F. Tawfic, and J. A. Mahmood, 1992, "The Transfer of ^{137}Cs from Soil to Plants," *Environment International*, 18:211-215.

Myttenaere, C. P. Bourdeau, and M. Masset, 1969, "Relative Importance of Soil and Water in the Indirect Contamination of Flooded Rice with Radiocesium," *Health Physics*, 16:701-707.

Neilsen, B. and M. Strandberg, 1988, "*A Literature Study of the Behaviour of Cesium, Strontium, and Plutonium in the Soil-Plant Ecosystem*," Risoe National Laboratory, DK-4000 Roskilde, Denmark, pp. 1-71.

Ng, Y.C., C. S. Colsher, and S. E. Thompson, 1979, "Transfer Coefficients for Terrestrial FoodchainXTheir Derivation and Limitations," CONF-7810179-1.

Nygren, P., P. Hari, T. Raunemaa, M. Kulmala, S. Luokkanen, M. Holmberg, and E. N. Nikinmaa, 1994, "Behaviour of ^{137}Cs from Chernobyl Fallout in a Scots Pine Canopy in Southern Finland," *Canadian Journal of Forest Research*, 24:1210-1215.

Price, K. R., 1973b, *Tumbleweed and Cheat Grass Uptake of Transuranium Elements Applied to Soil as Organic Acid Complexes*, BNWL-1755.

Routson, R.C, 1975, *The Effect of Soil Concentrations on the Tumbleweed Uptake of Sr-90 and Cs-137 from a Burbank Sand*, BNWL-1905.

Russell, R. S., 1966, "Radioactivity and Human Diet," ed. R. S. Russell, Pergamon Press, Oxford UK, Chapters 5 and 6, pp. 213-246.

Russell and Newbould, 1966,

Saar, R. A. and J.H. Weber, 1982, "Fulvic Acid: Modifier of Metal-ion Chemistry," *Environmental Science and Technology*, 16:510-517.

Sanchez, A. L., W. R. Schell, and E. D. Thomas, 1988, "Interactions of ^{57}Co, ^{90}Sr, and ^{137}Cs with Peat Under Acidic Precipitation Conditions," *Health Physics*, 54:317-324.

Schulz, R. K., R. Overstreet, and I. Barshad, 1960, "On the Soil Chemistry of Cesium137," *Soil Science* 89:16–27.

Seel, J. F., F. W. Whicker, and D. C. Adriano, 1995, "Uptake of ^{137}Cs in Vegetable Crops Grown on a Contaminated Lakebed," *Health Physics*, 68:793-799.

Sheppard, M. I., T. T. Vandergraaf, D. H. Thibault, and J. A. Keith Reid, 19883, "Technetium and Uranium: Sorption by and Plant Uptake from Peat and Sand," Health Physics 44:635–643, **S9**.

USNRC, 1979, *Regulatory Guide 1.109*, "Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I, Revision 1, U.S. Nuclear Regulatory Commission, Office of Standards Development, Washington D.C.

Van Bergeijk, K. E., H. Noordijk, J. Lembrechts, and M. J. Frissel, 1992, Influence of pH, Soil Type, and Soil Organic Matter Content on the Soil-to-Plant Transfer of Radiocesium and Strontium as Analyzed by a Nonparametric Model, *Journal of Environmental Radioactivity*, 15:265-276.